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Estimates of waterbird numbers wintering in Ireland, 2011/12 – 2015/16

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Annual monitoring of wintering waterbirds is carried out under the I-WeBS and WeBS schemes in the Republic and Northern Ireland respectively. These surveys are carried out from September to March each year, largely by a dedicated volunteer network, and are the principal tools used in the conservation of Ireland's wintering waterbirds and their wetland habitats. This study presents population estimates and 1% thresholds for wintering waterbirds in Ireland for the period 2011/12 to 2015/16 inclusive. Estimates were generated based on annual peak counts with imputation and include the results of more targeted surveys (i.e. goose and swan species censuses, non-estuarine surveys) where these improve the accuracy of estimates for the species in question. Estimates were generated for a total of 44 waterbird species, using data from 684 wetland sites across the Republic of Ireland and Northern Ireland. The total number of waterbirds estimated was 757,910, comprising 38% wildfowl (21 species), 6% wildfowl allies (8 species) and 57% waders (15 species). Total numbers have declined by 138,160 (15%) since the 2006/07-2010/11 period, with waders experiencing the largest declines; the combined totals of 15 wader species having declined by over 19%. Golden Plover *Pluvialis apricaria* and Lapwing *Vanellus vanellus* were the most numerous wader species recorded and Wigeon *Mareca penelope* and Teal *Anas crecca* were the most numerous wildfowl. Eight of the 44 species have increased by more than 5% since the previous estimates for 2006/07 – 2010/11, whereas 29 species declined by 5% over the same period. Many species are undergoing similar declines at flyway level, although the impact of local pressures and threats at Irish wetland sites should not be overlooked. Ireland continues to hold internationally important numbers of several waterbird populations, most notably Icelandic Whooper Swan *Cygnus cygnus,* Greenland White-fronted Goose *Anser albifrons flavirostris,* Greenland Barnacle Goose *Branta leucopsis,* East Canadian High-Arctic Light-bellied Brent Goose *Branta bernicla hrota,* Europe-wintering Great Northern Diver *Gavia immer,* North European Ringed Plover *Charadrius hiaticula,* Icelandic Black-tailed Godwit *Limosa limosa islandica* and North European/North Russian Bar-tailed Godwit *Limosa lapponica.*

Introduction

Waterbirds provide a number of important ecosystem services by acting as predators, herbivores, and as vectors of seeds, invertebrates and nutrients. In these roles they help maintain the diversity of other organisms, control pests and serve as effective bioindicators of the ecological condition of the wetlands they inhabit (Green & Elmberg 2013). These wetlands in turn provide hugely valuable services including

Plate 1. Black-tailed Godwit (Richard T. Mills).

water provision and purification, fixation of nutrient run-off, flood prevention, food production and carbon sequestration (Zedler & Kercher 2005). Conservative estimates of the value of these services from wetlands to the Irish economy run into the hundreds of millions (Bullock *et al.* 2008).

Ireland's location along the East Atlantic Flyway and proximity to major waterbird breeding areas in the Arctic, together with its mild climate and abundance of coastal and inland wetlands, make it a very important area for non-breeding wildfowl and waders during the winter months (Boland & Crowe 2012). Indeed, over 850,000 waterbirds were estimated to winter in Ireland during the last period assessed (Crowe & Holt 2013), and Ireland is important in a flyway context for several species including Light-bellied Brent Goose *Branta bernicla hrota*, Greenland White-fronted Goose *Anser albifrons flavirostris* and Whooper Swan *Cygnus cygnus,* amongst others. Regular reassessment of the relative importance of Ireland in a flyway context for wintering waterbird species is important and allows for the evaluation of the current importance of individual sites in a national and international context, facilitating site protection and management (Crowe & Holt 2013). To date, more than 100 wetlands in Ireland are designated as Special Protection Areas (SPA) under the E.U. Birds Directive (2009/147/EC) and 33 of these are also designated as Ramsar sites (under the 1971 Ramsar Convention on Wetlands).

These important data are underpinned by monitoring of waterbirds at our wetland sites. Selected species groups at some of the larger wetlands were first counted systematically as far back as the 1940s, but it wasn't until the 1970s that a comprehensive baseline survey was carried out. The Wetlands Enquiry (1971/72-1974/75; Hutchinson 1979) allowed numbers of most wintering wildfowl and wader species to be estimated for the first time and helped create a register of wetlands and their relative importance to wintering waterbirds. The Winter Wetlands Survey (1984/85-1986/87; Sheppard 1993) repeated and improved on the efforts of the previous decade and for the first time provided a trend for wintering waterbird numbers in Ireland.

Annual monitoring of waterbirds in Northern Ireland continued from the Winter Wetlands Survey and became what is now the UK Wetland Bird Survey (WeBS) (Delany 1996). The Irish Wetland Bird Survey (I-WeBS), modelled on its UK counterpart, began in winter 1994/95 (Delany 1996). Together the two schemes aim to monitor the numbers and distribution of waterbird populations wintering in Ireland across the long-term, enabling population numbers and trends of individual species to be established and updated on an ongoing basis. Since these monitoring schemes began, they have been the principal tools used in the conservation of Ireland's wintering waterbirds and the wetland habitats upon which they rely. Results have been used to provide population estimates and trends of waterbird species wintering on the island of Ireland during the periods 1994/95-1998/99, 1999/2000-2003/04 (Crowe *et al.* 2008) and 2006/07-2010/11 (Crowe & Holt 2013).

Here we present population estimates, population trends and 1% thresholds for regularly-occurring waterbird species wintering on the island of Ireland during the period 2011/12-2015/16. These updates are based on the results of annual I-WeBS and WeBS surveys, and additional surveys targeted at individual species and non-estuarine coastal sites.

Methods

Sources of data

Counts carried out under I-WeBS (Republic of Ireland) and WeBS (Northern Ireland), have been the primary source of data for wintering waterbirds on the island of Ireland since the mid-1990's. These surveys are carried out by a network of dedicated volunteer birdwatchers and professional staff of the schemes' partner organisations. Both surveys are based on scheduled monthly core counts each winter, from September to March inclusive. Counts are recommended in all seven months, although this is not often achieved. Emphasis is put on achieving monthly counts during the mid-winter period of November to February, when numbers of most species reach their peak. The importance of achieving good coverage in January in particular is stressed to counters, as these totals contribute to the International Waterbird Census (IWC) coordinated by Wetlands International (https://www.wetlands.org/). Counts are conducted on pre-determined dates to maximise synchrony and minimise any duplicate counts of flocks moving between or within sites.

The estimates of numbers and trends in relative abundance presented here were based largely on I-WeBS and WeBS core counts. The core count methodology is insufficient for surveying several species that feed regularly on grasslands away from wetland sites including swan and goose species, Golden Plover *Pluvialis apricaria*, Lapwing *Vanellus vanellus* and Curlew *Numenius arquata.* Furthermore, a large proportion of the populations of a variety of wader species, particularly Ringed Plover *Charadrius hiaticula,* Sanderling *Calidris alba,* Purple Sandpiper *Calidris maritima* and Turnstone *Arenaria interpres,* occur along non-estuarine coast which is not monitored during core counts. To better account for the numbers and relative abundance of these species, data from targeted surveys were integrated or used in place of core count data where available (Crowe *et al.* 2015, Fox *et al*. 2018, Hall *et al.* 2016, Lewis *et al.* 2017, Irish Brent Goose Research Group 2018, Doyle *et al.* 2018).

All waterbird species that are relatively widespread in Ireland were included in these analyses, and were grouped into wildfowl (29 species, including swans, geese and ducks, and their allies, defined here as divers, grebes, Cormorant *Phalacrocorax carbo*, herons and rails) and waders (15 species, including Oystercatcher *Haematopus ostralegus,* plovers and sandpipers). Elusive species, such as Water Rail *Rallus aquaticus,* Moorhen *Gallinula chloropus,* Jack Snipe *Lymnocryptes minimus,* Snipe *Gallinago gallinago* and Woodcock *Scolopax rusticola* which have a secretive and retiring nature, and marine species such as Long-tailed Duck *Clangula hyemalis* and Black-throated Diver *Gavia arctica,* which are difficult to survey from land, were excluded from these analyses. Introduced species, including Canada Goose *Branta canadensis* and Greylag Goose *Anser anser* (the naturalised population) have been excluded as there is no conservation requirement to define 1% thresholds for site assessment under the EU Birds Directive. Gulls and terns are not considered as they are not routinely counted during core counts, and their distributions are generally too widespread for adequate monitoring by these methods alone. The scientific names of all species included in these analyses are presented in Table1 and those for other species are given where first mentioned in the text.

Estimates of waterbird totals

Raw count data for the period under consideration (2011/12 – 2015/16) were first modelled using a multiplicative log-linear index model with site, year and month factors (after Underhill & Prŷs-Jones 1994 and using the UINDEX4 DOS executable programme). The resulting fitted values were then used to impute values where counts were missing (i.e. site not visited that month) or where a count was deemed to be 'low quality' (i.e. where the surveyor believed the count was markedly lower than the true number present).

Model considerations:

- In addition to data for the five-year period under consideration (2011/12-2015/16 inclusive), data from the two previous seasons (2009/10 and 2010/11) were included in the model to increase the pool of data and improve the model. These two years were then removed at the end to calculate the final five-year mean (for more detailed methodology please refer to Underhill & Prŷs-Jones 1994; Atkinson *et al.* 2006).
- The Underhill model was first run using all sites that had been surveyed in 50% or more of months during the seven-year period examined (Underhill & Prŷs-Jones 1994). This meant that the number of imputed counts in this first model was relatively low. The imputed values from this first run were then treated as actual counts during a second run of the model in which all sites were retained. This method ensured that sites with poor coverage did not impact on the imputed values for sites with good coverage (Crowe & Holt 2013).
- Counts were deemed to be of low quality where there was poor visibility, high disturbance levels, or because the site was only partially counted. For large wetland complexes with a number of count subsites, a count was identified as potentially incomplete if fewer than 75% of the subsites were surveyed, and less than 75% of the average number of birds present in previous years was recorded. Low quality counts were compared to fitted values for that month, and the larger of the two was used for analysis.

The resulting dataset was therefore complete for all months and seasons and comprised a combination of actual count data and imputed count data. All-Ireland estimates were then calculated using a five-year mean for the period 2011/12-2015/16 inclusive, consistent with the approach used previously in Ireland (Crowe & Holt 2013, Crowe *et al.* 2008) and the UK (Musgrove *et al.* 2011, Kershaw & Cranswick 2003, Rehfisch *et al.* 2003). This approach minimises the potential confounding effects of cold-weather movements (causing large-scale displacement) and disturbance (causing 'local' survey under- and overestimation).

Many wader species that winter in Ireland also have populations that occur here on passage in early autumn and late spring, but winter elsewhere (mostly further south in Continental Europe and/or Africa). To minimise inclusion of counts of passage populations, wader estimates were based on data from November to February each season only. This does not apply to passage populations of wildfowl populations in Ireland. The month with the highest value was selected (so that the index was based on maximum bird numbers), and all months with overlapping 90% consistency intervals (Underhill & Prŷs-Jones 1994) were also included.

Where available, the results of other targeted censuses (outlined above) were used in preference to I-WeBS/WeBS counts, as the species-specific targeting of survey effort is likely to provide better estimates of wintering numbers of these species. For those species which occur on non-estuarine coast, which is not routinely counted during I-WeBS or WeBS core counts, a median bootstrapped estimate from the third Non-Estuarine Waterbird Survey (NEWS-III) (Lewis *et al.* 2017) was added to the modelled counts. The updated bootstrap approach used for NEWS-III (Austin *et al.* 2017; Lewis *et al.* 2017) was retrospectively applied to NEWS-II data (Crowe *et al.* 2012) and subsequently to the previous set of Irish waterbird population estimates (Crowe & Holt 2013) to facilitate accurate calculation of short-term changes between the 2006/07-2010/11 and current periods.

Table 1. Flyway population estimates and trends of wildfowl and waders wintering in Ireland. All data based on African-Eurasian Migratory Waterbird Agreement (AEWA) Conservation Status Review 7 (CSR7) (AEWA 2018) (and published on wpe.wetlands.org).

Species	Population name	Flyway estimate	1% Flyway Threshold	Flyway trend
Mute Swan *Cygnus olor*	Ireland	9,130 [1]	100	Stable
Tundra Swan (Bewick's) *Cygnus columbianus bewickii*	*bewickii,* Western Siberia & NE Europe/North-west Europe	21,000	220	Decrease
Whooper Swan *Cygnus cygnus*	Iceland/UK & Ireland	34,000	340	Increase
Greenland White-fronted Goose *Anser albifrons flavirostris*	*flavirostris,* Greenland/Ireland & UK	20,529 [2]	190	Decrease
Greylag Goose *Anser anser*	*anser,* Iceland/UK & Ireland	93,720	980	Decrease
Barnacle Goose[3] *Branta leucopsis*	East Greenland/Scotland & Ireland	72,162	810	Decrease
Brent Goose (Light-bellied) *Branta bernicla hrota*	*hrota*, Canada & Greenland/Ireland	36,500	400	Decrease
Common Shelduck *Tadorna tadorna*	North-west Europe	250,000	2500	Stable
Wigeon *Mareca penelope*	Western Siberia & NE Europe/NW Europe	1,300,000-1,500,000	14000	Decrease?
Gadwall *Mareca strepera*	*strepera,* North-west Europe	110,000-138,000	1200	Increase
Eurasian Teal *Anas crecca*	*crecca,* North-west Europe	500,000	5000	Increase?
Mallard *Anas platyrhynchos*	*platyrhynchos*, North-west Europe	4,200,000-6,700,000	53000	Stable?
Northern Pintail *Anas acuta*	North-west Europe	65,000	600	Stable/ Fluctuating
Northern Shoveler *Spatula clypeata*	North-west & Central Europe (wintering)	60,000-70,000	650	Increase?
Common Pochard *Aythya ferina*	North-east Europe/ North-west Europe	200,000	2000	Increase
Tufted Duck *Aythya fuligula*	NW Europe (wintering)	800,000-1,000,000	8900	Decrease?
Greater Scaup *Aythya marila*	*marila,* Northern Europe/ Western Europe	150,000-275,000	3100	Decrease
Common Eider *Somateria mollissima*	*mollissima,* Baltic, Denmark & Netherlands	930,000	9,800	Stable/ Fluctuating
Common Scoter *Melanitta nigra*	*mollissima,* Norway & Russia W Siberia & N Europe/ W Europe & NW Africa	510,000-525,000 687,000-815,000	5,200 7500	Stable/ Stable/ Increase?
Common Goldeneye *Bucephala clangula*	*clangula,* NW & Central Europe (wintering)	1,000,000-1,300,000	11400	Stable/ Decrease
Red-breasted Merganser *Mergus serrator*	NW & Central Europe (wintering)	70,000-105,000	860	Stable/ Decrease?
Red-throated Diver *Gavia stellata*	North-west Europe (wintering)	216,000-429,000	3000	Stable?
Great Northern Diver *Gavia immer*	Europe (wintering)	5,100-6,300	50	Decrease?
Little Grebe *Tachybaptus ruficollis*	*ruficollis,* Europe & North-west Africa	375,000-597,000	4700	Stable/ Decrease?

Table 1 (Continued).

Species	**Population name**	**Flyway estimate**	**1% Flyway Threshold**	**Flyway trend**
Great Crested Grebe *Podiceps cristatus*	*cristatus*, North-west & Western Europe	513,000-764,000	6300	Stable/ Decrease?
Great Cormorant *Phalacrocorax carbo*	*carbo,* North-west Europe	127,500	1200	Decrease
Little Egret *Egretta garzetta*	*garzetta,* Western Europe, NW Africa	106,000-116,000	1100	Decrease
Grey Heron *Ardea cinerea*	*cinerea,* Northern & Western Europe	347,000-712,000	5000	Decrease
Common Coot *Fulica atra*	*atra*, North-west Europe (wintering)	1,200,000-2,000,000	15500	Stable/ Decrease?
Eurasian Oystercatcher *Haematopus ostralegus*	*ostralegus*, Europe/South & West Europe & NW Africa	850,000-950,000	8200	Stable/ Decrease?
Ringed Plover *Charadrius hiaticula*	*hiaticula,* Northern Europe/ Europe & North Africa	47,000-62,000	540	Decrease/ Stable
European Golden Plover *Pluvialis apricaria*	*altifrons*, Iceland & The Faroes/ East Atlantic coast	930,000	9300	Uncertain
Grey Plover *Pluvialis squatarola*	*squatarola*, W Siberia/W Europe & W Africa	200,000	2000	Decrease
Northern Lapwing *Vanellus vanellus*	Europe, W Asia/Europe, N Africa & SW Asia	5,500,000-9,500,000	72300	Decrease
Red Knot *Calidris canutus*	*islandica,* NE Canada & Greenland/Western Europe	500,000-56,5000	5300	Stable/ Fluctuating
Sanderling *Calidris alba*	*alba,* East Atlantic Europe, West & Southern Africa (wintering)	200,000	2000	Stable
Purple Sandpiper *Calidris maritima*	N Europe & W Siberia (breeding) NE Canada & N Greenland (breeding)	50,000-100,000 11,000	710 110	Increase Decrease
Dunlin *Calidris alpina*	*alpina,* NE Europe & NW Siberia/ W Europe & NW Africa	1,330,000	13300	Stable/ Decrease?
Black-tailed Godwit *Limosa limosa*	*islandica,* Iceland/ Western Europe	98,000-134,000	1100	Increase
Bar-tailed Godwit *Limosa lapponica*	*lapponica,* Northern Europe/ Western Europe	150,000	1500	Increase
Eurasian Curlew *Numenius arquata*	*arquata,* Europe/Europe, North & West Africa	637,000-876,000	7600	Decrease?
Common Greenshank *Tringa nebularia*	Northern Europe/SW Europe, NW & West Africa	230,000-470,000	3300	Stable/ Increase?
Common Redshank *Tringa totanus*	*totanus,* Britain & Ireland/Britain, Ireland, France	76,500	760	Decrease
	robusta, Iceland & Faroes/ Western Europe	150,000-420,000	2400	Decrease?
Ruddy Turnstone *Arenaria interpres*	NE Canada & Greenland/ W Europe & NW Africa	100,000-200,000	1400	Increase

[1] Mute Swan flyway estimate based on current study.
[2] Greenland White-fronted Goose flyway estimate from Fox *et al.* 2018.
[3] Barnacle Goose flyway estimate and trend from Mitchell & Hall 2018; 1% flyway threshold taken from AEWA 2018

Plate 2. Black-tailed Godwits, Dunlin, Redshank and Knot (John Fox).

Results

Coverage

In total, 631 wetland sites were visited in the Republic of Ireland in the five seasons between 2011/12 and 2015/16 inclusive. This total includes the majority of the key wetland sites that are of significant importance for birds (Burke *et al.* 2018) and is predominated by lakes and turloughs (58%), estuaries (11%), rivers and callows (13%) and non-estuarine coast (5%). The remaining 13% of sites include reservoirs, marshes and bogs, quarries and gravel pits, and grasslands. The number of sites covered in any one year ranged between 261 and 234, with between 78% and 89% of sites covered in January in any one season.

In Northern Ireland, 53 wetland sites were visited in the five seasons from 2011/12 to 2015/16. These were mostly lakes and turloughs (55%), estuaries (23%) and reservoirs (17%), with marshes, rivers and grasslands making up the remaining 5%. The number of sites covered in any one year ranged between 29 and 41, with between 71% and 90% of sites covered in January in any one season. The majority of key wetlands that are of significant importance for waterbirds (Frost *et al.* 2017) were counted during this period.

Estimates

Estimates were generated for a total of 44 species. The total number of waterbirds estimated was 757,910, comprising 38% wildfowl (21 species), 6% wildfowl allies (8 species) and 57% waders (15 species) (Table 2a, 2b). The estimates generated for 15 species must be treated as conservative as they are widely distributed in a variety of wetland and non-wetland habitats that are under-sampled during I-WeBS, WeBS and the targeted surveys. Golden Plover and Lapwing were the most abundant species recorded overall, estimated at 92,060 and 84,690 individuals respectively, collectively comprising 23% of the total number of waterbirds estimated. Wigeon *Mareca penelope* (55,730), Teal *Anas crecca* (35,740) and Light-bellied Brent Goose *Branta bernicla hrota* (35,150) were the most numerous wildfowl species, collectively comprising 17% of total waterbirds. Of the waders, Oystercatcher, Dunlin *Calidris alpina* and Curlew were also relatively numerous (>30,000 individuals).

Changes in estimates

The estimates for Bewick's Swan *Cygnus columbianus bewickii* and Scaup *Aythya marila* showed declines of greater than 50% since the previous estimate, at 74% and 58% respectively (Table 2a). The numbers of eight species declined by between 25% and 50% since the previous period: Red-throated Diver *Gavia stellata,* Knot *Calidris canutus,* Goldeneye *Bucephala clangula,* Common Scoter *Melanitta nigra,* Purple Sandpiper, Shoveler *Spatula clypeata,* Pochard *Aythya farina* and Great Crested Grebe *Podiceps cristatus* (Table 2a, 2b). The estimates for Common Scoter, Red-throated Diver and Great Crested Grebe should be treated with caution however, as these are conservative estimates for species that can be distributed a considerable distance offshore and therefore are under-recorded through core count methodology. In total, 27 waterbird species underwent declines of greater than 10% since the last period, including those mentioned above. In contrast, a total of seven species showed increases of greater than 10%: Eider *Somateria mollissima* (76%), Grey Heron *Ardea cinerea* (24%), Gadwall *Mareca strepera* (17%), Greenshank *Tringa nebularia* (17%), Great Northern Diver *Gavia immer* (15%), Whooper Swan (13%) and Sanderling (13%).

Table 2a. All-Ireland estimates and trends in relative abundance of wildfowl and their allies. The estimates given are based on mean of peak counts with imputation unless otherwise stated. The 1% thresholds are calculated (with roundings) as 1% of the respective all-Ireland totals. Percentage changes in estimates since the previous period (2006/07-2010/11; Crowe & Holt 2013, as revised) and the first set of I-WeBS/WeBS estimates (1994/95-1998/99; Crowe *et al.* 2008).

Species	**Species code**	**Republic of Ireland [1]**	**Northern Ireland [1]**	**All-Ireland [1]**	**All-Ireland 1% threshold [1]**	**% NEWS**	**% Change Estimates**	
							Short-term (2006/07-10/11)	**Long-term (1994/95-98/99)**
Mute Swan*	MS	7,032	2,094	9,130	90	2.0	-0.9	-24.9
Bewick's Swan [2]	BS	21	0	20	20	-	-73.8	-98.6
Whooper Swan [2]	WS	11,852	3,518	15,370	150	-	13.4	39.6
Greenland White-fronted Goose [2]	NW	9,500	87	9,590	100	-	-14.5	-20.9
Greylag Goose	GJ	1,954	1,598	3,550	35	-	-20.8	-20.8
Barnacle Goose [2]	BY	16,237	0	16,240	160	-	-7.2	101.1
Light-bellied Brent Goose [2]	PB	-	-	35,150	350	-	-15.5	96.1
Shelduck	SU	6,378	3,783	10,160	100	1.2	-14.2	-30.4
Wigeon*	WN	50,452	5,282	55,730	560	3.3	-12.0	-37.6
Gadwall	GA	515	377	890	20	0.0	17.1	34.8
Teal*	T.	27,644	8,096	35,740	360	6.6	6.2	-21.6
Mallard*	MA	18,810	9,423	28,230	280	7.5	-4.6	-41.2
Pintail	PT	1,017	557	1,570	20	0.0	-12.8	-4.8
Shoveler	SV	1,865	150	2,020	20	0.1	-30.6	-32.9
Pochard	PO	4,729	6,422	11,150	110	0.0	-30.4	-77.3
Tufted Duck	TU	16,927	10,544	27,470	270	>0.0	-11.1	-34.0
Scaup	SP	167	2,485	2,650	25	>0.0	-57.9	-58.2
Eider*	E.	1,373	4,288	5,660	55	25.6	76.3	100.7
Common Scoter*	CX	10,607	34	10,640	110	22.6	-31.5	-42.8
Goldeneye	GN	1,256	2,559	3,820	40	0.3	-36.8	-67.8
Red-breasted Merganser	RM	1,913	519	2,430	25	31.7	1.3	-33.6
Red-throated Diver*	RH	657	109	770	20	53.1	-42.5	-38.6
Great Northern Diver*	ND	2,128	110	2,240	20	68.1	14.9	-
Little Grebe*	LG	1,594	601	2,200	20	4.9	-1.3	-16.3
Great Crested Grebe*	GG	1,734	1,195	2,930	30	3.6	-28.2	-42.9
Cormorant*	CA	7,967	2,907	10,870	110	33.5	-21.1	-15.3
Little Egret	ET	1,274	117	1,390	20	11.5	-2.8	-
Grey Heron*	H.	1,943	662	2,610	25	30.5	24.3	-5.1
Coot	CO	13,303	5,216	18,520	190	0.0	-16.7	-34.6
Total wildfowl				**287,210**				
Total wildfowl allies				**41,530**				

* These estimates must be treated as conservative on the basis that they are widely distributed in a variety of wetland and non-wetland habitats that are undersampled during I-WeBS, WeBS and the special surveys, such as on large, small and ephemeral wetlands, or considerable distances offshore and not detected during counts from land-based vantage points.

[1] Estimates were derived by summing the core counts with those from the 2015/16 Non-estuarine Waterbird Survey (NEWS III) (Lewis *et al.* 2017) and have been rounded up or down to the nearest '10'. A minimum 1% threshold of 20 has been applied to all species with totals less than 2000, and remaining thresholds have been rounded as follows: 21-100 to the nearest five; 101-1000 to the nearest ten.

[2] Estimates from targeted censuses apply, including for swans (Crowe *et al.* 2015, Hall *et al.* 2016), Greenland White-fronted Goose (Fox *et al.* 2018), Barnacle Goose (Doyle *et al.* 2018) and Light-bellied Brent Goose (Irish Brent Goose Research Group 2014, Colhoun *et al.* 2015, 2017, 2018). Data from Brent Goose census in October 2017 is not representative of distribution later in the winter when majority winter in ROI.

Table 2b. All-Ireland estimates and trends in relative abundance of waders. The estimates given are based on mean of peak counts with imputation unless otherwise stated. The 1% thresholds are calculated (with roundings) as 1% of the respective all-Ireland totals. The percentage change in estimates since the previous period (2006/07-2010/11; Crowe & Holt 2013 as revised) are given, together with the % change since 1994/95-1998/99 (Crowe *et al.* 2008).

Species	Species code	Republic of Ireland [1]	Northern Ireland [1]	All-Ireland [1]	All-Ireland 1% threshold [1]	% NEWS	% Change in Estimates	
							Short-term (2006/07-10/11)	Long-term (1994/95-98/99)
Oystercatcher	OC	42,875	17,665	60,540	610	30.0	-21.2	-7.9
Ringed Plover	RP	10,545	1,113	11,660	120	54.5	-16.5	-18.7
Golden Plover*	GP	80,707	11,357	92,060	920	4.5	-23.5	-43.6
Grey Plover	GV	2,812	131	2,940	30	7.0	-5.8	-54.3
Lapwing*	L.	69,823	14,863	84,690	850	6.4	-16.4	-67.2
Knot	KN	13,752	2,520	16,270	160	3.1	-42.2	-43.3
Sanderling	SS	7,572	849	8,420	85	44.4	13.2	34.9
Purple Sandpiper	PS	465	197	660	20	74.9	-31.3	-80.7
Dunlin	DN	37,409	8,350	45,760	460	13.3	-23.2	-61.6
Black-tailed Godwit	BW	17,862	1,933	19,800	200	0.6	4.2	44.9
Bar-tailed Godwit	BA	13,385	3,147	16,530	170	4.0	3.9	6.5
Curlew*	CU	28,300	6,938	35,240	350	30.3	-13.4	-42.3
Greenshank	GK	1,208	109	1,320	20	32.0	16.8	11.9
Redshank	RK	16,812	6,988	23,800	240	16.4	-23.6	-19.2
Turnstone	TT	6,296	3,180	9,480	95	49.0	-20.6	-28.0
Total waders				**429,170**				
Total waterbirds				**757,910**				

Footnotes are given under Table 2a.

Data from NEWS-III collected in the winter of 2015/16 (Lewis *et al.* 2017) accounted for over 11% of the waterbirds estimated overall. Over 50% of the Purple Sandpiper (75%), Great Northern Diver (68%), Ringed Plover (55%) and Red-throated Diver (53%) estimates were recorded during NEWS-III (Table 2a, 2b), as was 25-50% of the populations of nine species and 10-25% of another four species. Most of these were waders but also included wildfowl and ally species. The most numerous wader species recorded during NEWS-III was Oystercatcher, with 18,133 individuals estimated, which represents 30% of their all-Ireland population estimate (Table 2b). Declines along non-estuarine coast shown between NEWS-II in 2006/07 (Crowe *et al.* 2012) and NEWS-III in 2015/16 were responsible for a significant part of the decrease in estimates of Red-throated Diver, Common Scoter, Purple Sandpiper, Ringed Plover and Turnstone. Conversely, increases along non-estuarine coast between NEWS-II and NEWS-III were largely responsible for increases in estimates for Eider, Grey Heron, Greenshank and Great Northern Diver.

Discussion

Annual monitoring of waterbirds in Ireland through I-WeBS and WeBS, together with additional more targeted surveys, allow for the calculation of robust population estimates and trends for wintering waterbirds on the island of Ireland on a regular basis. The results presented here for the period 2011/12–2015/16 are the fourth such set of estimates published, updating those from 2006/07-2010/11 (Crowe & Holt 2013). The estimates for 44 species are given, resulting in a total of 757,910 waterbirds – a decline of 138,160 (15%) since the 2006/07-2010/11 period. The combined totals of the 15 wader species examined declined by over 102,310 birds (19%) since the previous period, while wildfowl and wildfowl ally numbers have declined by over 28,000 (9%) and 7,600 (16%) respectively. Total numbers of wildfowl and waders wintering in Ireland have shown a continued decrease through each set of estimates published (Crowe *et al.* 2008, Crowe & Holt 2013). The first such set of estimates published for the period 1994/95-1998/99 estimated 1,255,575 waterbirds across 42

Plate 3. Dunlin and Black-tailed Godwit (Richard T. Mills).

species, meaning a decline of 40% of our total waterbird population has occurred in the intervening 17 years.

Ireland continues to host a very high proportion of the flyway populations of East Canadian High-Arctic breeding Light-bellied Brent Goose (98% of flyway population winters in Ireland), Greenland White-fronted Goose (47%), Icelandic Whooper Swan (45%), Europe-wintering Great Northern Diver (45%), Greenland Barnacle Goose *Branta leucopsis* (23%) and North European Ringed Plover (22%). Wintering numbers of Black-tailed Godwit *Limosa limosa* (18% of flyway population) and Bar-tailed Godwit *Limosa lapponica* (11%) were also highly important in a flyway context (>10% of flyway population), though the relative proportion of Black-tailed Godwits wintering here has declined by 13% (previously 31%, Crowe & Holt 2013). The island of Ireland has a closed population of Mute Swans *Cygnus olor*, and the current estimate is almost identical to that of the previous period, though that estimate itself represented a 20% decline since the late 1990's and early 2000's (Crowe *et al*. 2008).

The majority of wildfowl and ally species were present in lower numbers than in the previous period. Only four wildfowl species showed increases greater than 5%. The largest increase was for Eider (76%), although given the limitations of core count methodology in recording this coastal species and the fact that the populations are declining at flyway level, comparison of estimates here is almost certainly unreliable. Gadwall showed a 17% increase, although the Irish population is still relatively small at <900 birds. Whooper Swan numbers in Ireland increased between the 2010 and 2015 censuses (13%), though at a slower rate than elsewhere in their wintering range (Hall et al. 2016), with increases in the Republic of Ireland (13%) but a 24% reduction in Northern Ireland. Finally, Teal numbers increased by 6% on the previous period. Amongst the allies, both Grey Heron (24%) and Great Northern Diver (15%) increased by more than 5%, though as with Eider these species are not well covered by I-WeBS core counts.

By contrast, 18 species of wildfowl and allies underwent declines of over 5%. A lot of these species are also declining at flyway level. In many cases it is likely that climate change is a significant factor in the species returning to Ireland in reduced numbers, irrespective of flyway trends. For example,

Plate 4. Pintail (Michael Finn).

only 21 Bewick's Swan returned to Ireland (Wexford) in recent years, and this species has declined by 74% since the previous period. The decline of Bewick's Swans in Ireland pre-dates their decline at international level (Kennedy *et al.* 1954, Ruttledge 1966, Sheppard 1993) and has been attributed to milder conditions closer to the breeding grounds and the species 'short-stopping' elsewhere as a result (Crowe *et al.* 2005, Worden *et al.* 2006). It is likely that this species will cease to be a regular wintering species here in the near future. The same cause is probably a factor in other species wintering here in reduced numbers, as the migration as far as Ireland is becoming increasingly disadvantageous rather than necessary. Recent research by Pavón-Jordán *et al.* (2018), which included both I-WeBS and WeBS data, has provided evidence for a long-term north-eastwards shift of the centre of the wintering population of species preferring deep waters (e.g. Pochard, Tufted Duck *Aythya fuligula,* Goldeneye, Red-breasted Merganser *Mergus serrator,* Coot *Fulica atra*, Cormorant, Great Crested Grebe) and changing shifts in the centre of the wintering population of shallow-water species (e.g. Pintail *Anas acuta*, Shoveler, Teal, Wigeon, Mallard *Anas platyrhynchos*, Gadwall, Shelduck *Tadorna tadorna*) in response to large-scale changes in winter weather conditions (linked to NAO index values). Lehikoinen *et al.* (2013) and Fox *et al.* (2016) found similar evidence for Tufted Duck and Goldeneye, and Wigeon, respectively.

The decline of the Greenland White-fronted Goose has been due to poor breeding productivity, with the population now standing at around half of what it was at its recent peak in 1999/2000. In Ireland, numbers in Wexford have remained relatively stable as birds abandon former sites around the country and the species becomes increasingly concentrated on the Slobs and neighbouring sites (Fox *et al.* 2018, Weegman *et al.* 2016). Around 80% of the Irish population of Greenland White-fronts now winter at a small number of sites in Wexford (Fox *et al.* 2018). Although the North Slob is managed for these geese, their continual loss of range in the north and west of the country puts the Irish population at increased risk of any potential stochastic or longer-term threats at those few sites in Wexford.

Total wader numbers in Ireland have declined almost 20% since the previous estimates 5 years ago. Numbers of Greenshank (17%) and Sanderling (13%) have showed significant increases in the last 5 years and are either stable or increasing at flyway level. Black-tailed Godwits are increasing internationally and showed modest gains (4%) in Ireland. Declines of >20% are evident for Knot, Purple Sandpiper, Redshank, Golden Plover, Dunlin, Oystercatcher and Turnstone, with 10% declines for Ringed Plover, Lapwing and Curlew. Golden Plover, Lapwing, Oystercatcher and Dunlin have long been among the most numerous waterbird species in Ireland (Crowe *et al.* 2008, Crowe & Holt 2013), so these large and rapid declines will have a disproportionately negative effect on the total waterbird numbers at many sites. It is also worth noting that the number of wetlands regularly supporting 20,000+ migratory waterbirds, one of the thresholds for international site importance under the Ramsar Convention, has fallen considerably since I-WeBS began. Fifteen sites in the Republic of Ireland supported over 20,000 wintering waterbirds in 2004/05, when the first set of population estimates were published (Boland & Crowe, 2006; Crowe *et al.* 2008), falling to nine in 2010/11 when the last set of estimates were published (Crowe *et al.* 2012; Crowe & Holt 2013), and only five sites met the criteria in the recent period from 2011/12 to 2015/16 (Burke *et al.* 2018).

As with wildfowl, there is an increasing body of evidence linking changes in temperature and climate to easterly shifts in wintering distribution for species such as Lapwing, Golden Plover (Gillings *et al.* 2016), Ringed Plover, Dunlin, Knot, Redshank, Grey Plover *Pluvialis squatarola* and Oystercatcher, amongst others (Austin & Rehfisch 2005). Many of the wader species declining in Ireland are also decreasing at flyway level, suggesting causative factors further afield than Ireland. Kubelka *et al.* (2018) found that levels of predation of shorebird nests in the Arctic have increased threefold in the last 70 years, with 70% of total nests now being depredated. Higher rates of predation were associated with increased temperatures, indicating that climate change is affecting total numbers of wild bird populations as well as causing the shifts in range discussed above. However, this should not mask the many local pressures faced by wintering waterbirds. In Ireland, many waterbirds are vulnerable to recreational disturbance, habitat modification and loss, and potential impacts from increased aquaculture and renewable energy developments, each of which has the potential to lower survival rates and total numbers of their respective Irish and flyway populations as a result.

It is clear that there are a wide range of anthropogenic and environmental factors that are affecting waterbird populations; likely underpinning the observed declines at national and flyway level. Because waterbirds are largely migratory, there are considerable challenges in addressing these declines as many different factors may affect populations on the breeding grounds, along their migratory routes and in the wintering areas. An integrated temporal and spatial approach is required to address waterbird declines along the east Atlantic flyway and further afield, which will undoubtedly be confounded by differing economic jurisdictions and priorities. Despite calls for international, collaborative and strategic measures to address these declines for many years (e.g. Stroud *et al.* 2006) and various initiatives put in place, there are few signs of improving trends for many species. Continued monitoring is therefore important to keep track of our changing waterbird populations as well as to input into international programmes such as the International Waterbird Census and related research, to ultimately inform future conservation policies.

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The status and ecology of a remnant population of Ring Ouzel *Turdus torquatus* in the MacGillycuddy's Reeks, Kerry

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Keywords: Ring Ouzel, *Turdus torquatus,* MacGillycuddy's Reeks, population decline, upland habitat, Kerry

The Ring Ouzel *Turdus torquatus* is one of the most poorly studied and threatened bird species in Ireland. The species has suffered a precipitous decline in population size and range with breeding now apparently largely confined to just two counties, Kerry and Donegal. Population changes in Ireland appear to mirror widespread declines across the species' range in Britain. A study of Ring Ouzels in the MacGillycuddy's Reeks was initiated in 2008 to determine the status of the ouzel population in Kerry. Potential breeding habitat was identified and efforts made to visit all sites to locate birds. Where birds were detected and visually located, habitat characteristics were assessed. The number of apparently occupied sites declined from eleven in 2008 to four by 2011 although some early singing males in 2008 may have been passage migrants. At least three 'core' sites remained occupied between 2012-2017. Ouzels were largely confined to high elevation sites (400-850m a.s.l.) on steep-vertical slopes with extensive rock and boulders. Song peaked in early morning (0610-0930hrs) and declined thereafter. Males sang from arrival up until at least late June suggesting that some pairs may be double-brooded. Observations suggest a possible association with wet flushes as foraging sites within short distances of nests. Ring Ouzels are at risk of extinction in Ireland. A comprehensive survey of core areas in Kerry and Donegal is urgently needed. Habitat management to restore or maintain heather-grass mosaics at key sites should be an important and urgent conservation measure for Ring Ouzels in Ireland.

Introduction

The Ring Ouzel *Turdus torquatus* is one of the most poorly studied and understood breeding species in Ireland. The species breeds generally above 300m across northern and central Europe from Ireland to Fenno-Scandanavia and north-western Russia and and Spain east to Turkmenistan and Iran (Cramp & Simmons 1998, del Hoyo *et al.* 2005). In continental Europe Ring Ouzels are birds of mountain steppe, including open coniferous forest, conifer-beech woodland, alpine scrub, heath and subalpine meadows above the tree line (del Hoyo *et al.* 2005). The nominate northern Fenno-Scandia subspecies *T. torquatus torquatus* (including the Irish and British populations) is believed to winter mainly in the Atlas Mountains of north-west Africa (del Hoyo *et al.* 2005). In Ireland and Britain, Ring Ouzels typically inhabit uplands up to 1,200m but also down to near sea level in parts of North

Plate 5. Ring Ouzel (Peter Curran).

Scotland and North-west Ireland (Ussher & Warren 1900, Sim *et al.* 2010). In Ireland as in Britain, at least in recent times, Ring Ouzels have been birds of steep crags, scree slopes and ravines in mountainous areas with varying degrees of heather cover (Sim *et al.* 2010). Migrants tend to arrive back from their wintering grounds at the end of March and early April. A small number of birds overwinter although this is apparently rare (Balmer *et al.* 2013).

In Britain birds typically nest from mid-April with egg laying peaking in late April-early May (Sim *et al.* 2010, 2012). Birds may relay following nest failure or have second nests after initial broods fledge: >50% of pairs in the North York Moors, England (Hutchinson & Fairbrother 2017) and 70% of females in Glen Clunie, Scotland (Sim *et al.* 2012). Nests in Ireland and Britain appear to be almost always on the ground in cliffs, crags and gullies, but are often found in trees in Europe (del Hoyo 2005, Sim *et al.* 2010) The diet of nestlings in Britain is dominated by invertebrates especially earthworms (Lumbricidae), but also adult and larval ground beetle (Carabidae) leatherjacket larvae (Tipulidae) and other insects (Burfield 2002, Buchanan *et al.* 2006). In contrast, adults and fledged juveniles switch to foraging on berries, especially Bilberry *Vaccinium myrtillus* and Crowberry *Empetrum nigrum,* once available in Scotland (Sim *et al.* 2012) and Rowan *Sorbus aucuparia* elsewhere in Europe (del Hoyo 2005). Juniper *Juniperus communis* berries appear to be the predominant diet of wintering birds in the Atlas Mountains in Morocco and elsewhere in N Africa (del Hoyo *et al.* 2005, Ryall & Briggs 2006).

Ring Ouzels were thought to be a widespread and reasonably common breeding bird in appropriate habitat in Ireland in recent historical times. Thompson (1849) cites records of the species from the hills west of Belfast, "every one (glens or ravines) of which boasted a pair or more of these birds", the Glens of Antrim, Donegal, Down (especially the Mournes), Carlingford Mountain in Louth, Achill Head in Mayo, the Clare hills, the Dublin and Wicklow mountains, the Slieve Aughties in Galway, Connemara, Slievenamon in Tipperary and the Comeraghs in Waterford. Thompson (1949) also noted that the Ring Ouzel "is common in the most rocky parts of the mountains of Kerry... in the same haunts with choughs and eagles". Ussher & Warren (1900) remark that the bird was found in all counties except Meath, Westmeath, Longford and Armagh but most notably "the higher mountains of Kerry, Waterford, Tipperary, Wicklow, Galway, Mayo, Sligo, Leitrim, Donegal and Down". A considerable decline was noted by the mid-20th century with the species having disappeared from many former haunts as well as being scarce at sites where it was still present (Kennedy *et al.* 1954). Declines in the Irish and British Ring Ouzel population were evident by the time of the first *Breeding Atlas* in 1968-72 (Sharrock 1976) and continued during the next four decades (Gibbons *et al.* 1993, Wotton *et al.* 2002, Sim *et al.* 2010). The most recent 2007-11 *Breeding Atlas* (Balmer *et al.* 2013) revealed a 57% decline in the species breeding range in Ireland since 1968-72 with breeding confirmed in Donegal and Kerry only, mirroring declines (-47%) in Britain (Figure 1). At least one and possibly up to three pairs of Ring Ouzel still bred in the Mourne Mountains in the period 1986-95 (A. McGeehan pers. comm). Wotton *et al.* (2002) considered the species was likely extinct as a breeding bird in Northern Ireland by 1999 although a pair was since recorded at a site in Derry and birds were reported from the same area in the early 1980s. A survey for the species in Donegal in 2002 located birds at 10 sites with possibly up to 15 pairs breeding (Cox *et al.* 2002). However, upland bird surveys in Sligo, Leitrim, north-west Cavan and north Mayo in 2003 did not locate any Ring Ouzels although a pair was located in Donegal (Cummins *et al.* 2003).

The Ring Ouzel has been listed as a species of high conservation concern in Ireland due to its severely declining national population (>80%) and an extremely restricted and evidently contracting range (Colhoun & Cummins 2013). Habitat change, loss of heather cover, low adult and juvenile

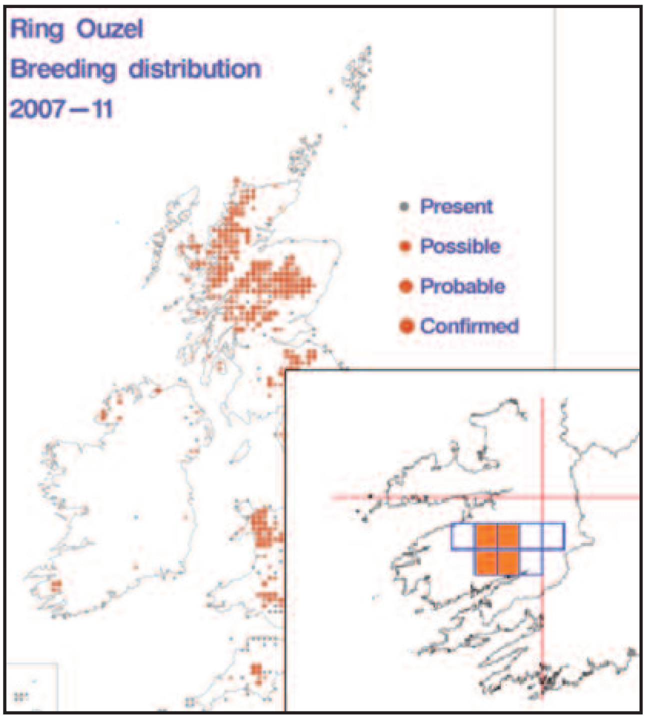

Figure 1. Distribution and breeding status of Ring Ouzel in Ireland in 2007-2011 (Balmer *et. al.* 2013). Inset shows range of the MacGillycuddy's Reeks breeding population (filled 10km squares) and previous breeding range (unfilled 10km squares) in the 1988-91 breeding Atlas (Gibbons et al. 1993). Map reproduced from *Bird Atlas 2007–11*, which is a joint project between BTO, BirdWatch Ireland and the Scottish Ornithologists' Club. Map reproduced with permission from the British Trust for Ornithology.

Figure 2. Ring Ouzel survey areas in the MacGillycuddy's Reeks and Mangerton Mountain, Kerry. Occupied sites are indicated by green triangles. No occupied sites were located in the Mangerton area. Inset shows the location of the survey area in Ireland.

survival rates, inter-specific competition with resident thrush species, habitat loss on wintering grounds, hunting pressure on migration and climate change have all been cited as driving declines (Burfield 2002, Buchanan *et al.* 2003, Beale *et al.* 2006, Ryall & Briggs 2006, Sim *et al.* 2010, 2011, Wotton *et al.* 2002). The main aim of this study was to locate potentially breeding Ring Ouzel in the MacGillycuddy's Reeks, County Kerry, in order to obtain an accurate estimate of the population size and range. As relatively little is known about the behaviour and habitat preferences of Irish Ring Ouzels, apart from the historical information (e.g. Thompson 1850, Ussher & Warren 1900) and some more recent survey work (Carruthers 1998, Cox *et al.* 2002), an additional aspect of the survey work was to investigate the ecology of the ouzel population and aspects of its breeding biology.

Methods

The study area was the MacGillycuddy's Reeks range and outlying mountains in the Mangerton area, situated south and west of Killarney, County Kerry (Figure 2). Ring Ouzels were surveyed during the breeding season (April-July) from 2008 to 2017. All sites occupied in previous years were revisited at least twice annually in years 2009 to 2011. Not all sites were visited in all years between 2012 and 2017. The Reeks study area covered some 52km^2 in area with much of the range over 800 m.a.s.l including Ireland's highest peak Corrán Tuathail (1,039 m). The main summit ridge is composed of a series of sharp arêtes and a broad summit plateau, with the north side of the Reeks marked by a series of coums (corries), small lakes and moraines. On the east side, the peaks of Purple Mountain (832 m) and Tomies (735 m) rise steeply from the of the Gap of Dunloe with extensive scree and boulder slopes. The Mangerton area is a broad plateau with its highest point at 839m. Much of the Reeks is dominated by upland grassland, heath and blanket bog.

Potential Ring Ouzel habitat was identified from aerial imagery and Ordnance Survey maps. Survey routes were designed to cover the entire study area and give the best chance of detecting singing birds. As Ring Ouzels are highly vocal their presence is best detected by song given suitable weather conditions. The survey followed recommended, previously described methods (Gilbert *et al.* 1998, Wotton *et al.* 2002) but were adapted because of the extremely steep nature of the terrain in the main Reeks study area. Conventional methods specify the use of transects 500 m apart

Plate 6. Male Ring Ouzel in flight near breeding site (site 10) in the MacGillycuddy's Reeks, Co. Kerry (Anthony McGeehan).

so that all areas within a tetrad (2 km^2) are covered to within 250 m. As Ring Ouzels are known to be especially vocal in the morning, each survey transect was completed within four hours of sunrise (e.g. 0550-0950hrs on 1 May) although additional time was spent observing occupied sites and nest sites after this time during the breeding season. Some of the more accessible sites where birds were recorded were revisited late in the day, within two hours of sunset, as thrushes, including Ring Ouzels, are known to be vocal at dusk (Cramp & Simmons 1998).

Transects were walked on both side of the main ridge with the aim of covering all areas within a tetrad to within 500m. The location of start, finish, and points along survey routes were plotted using a hand-held Global Positioning Systems (GPS) device. Every 500 m along the survey route, beginning with the initial start point, Ring Ouzel song was listened for over a minimum 10-minute period. After the first 5 minutes, if no bird was heard, a recording of ouzel song and calls was played for 20 – 30 seconds and repeated three times (Gilbert *et al.* 1998, Wotton *et al.* 2002). Following this a response (song or calls) was listened for over a five-minute period. Although playback of Ring Ouzel song was used on transect routes during initial surveys in 2008, no audible responses were detected and playback was not used subsequently (NPWS licence required under 35 1(d) of Wildlife Act). When Ring Ouzel presence was detected (song, calls or direct observation) every effort was made to locate individuals visually and plot their location (Irish grid) as accurately as possible on a map (OSI MacGillycuddy's Reeks 1:25,000) and/or using a GPS device where observation at close range was possible. This was often difficult for birds singing at a distance from steep cliffs. Ring Ouzel often move or change position between song bursts making accurate location difficult if birds are not located visually. Ring Ouzel nest sites, where located, were observed from a minimum distance of 500m to minimise any potential disturbance or affect behaviour.

Where Ring Ouzels were located at least 30 minutes was spent at each point observing birds (usually singing males), recording their behaviour and the habitat type used (Fossitt 2000). As well as the vegetation types where ouzels were located, habitat variables were quantified visually within 50 m and 500 m of the birds to give an estimate of the habitat characteristics used by ouzels (50 m) and those habitat types available in the wider area (500 m). Habitat characteristics included percentage cover of heather, grasses, boulder, scree, rock, and open water; distance to open water; number of trees; and distance to nearest tree. Slope and aspect were also recorded as well as the number of sheep within 50/500m as an index of grazing pressure (Fuller & Gough 1999).

Results

Occupancy

The number of occupied sites in the core Reeks survey area, based on the number of singing males, declined from 11 sites in 2008 to just four sites in 2011 (Table 1). However, all sites were visited just once in 2008, thus it is not known if all birds were resident breeders, at least two visits are required to determine occupancy. Only two sites were occupied consistently in all years although a third site (site 9) was occupied in most years and may have been missed in 2009. Active nests were located at only two sites (sites 10, 11 in Figure 2) and birds appeared to use the same nest sites in subsequent years. No ouzels were located at sites in the Mangerton area or at other sites outside the core Reeks area known to have held ouzels in the past (Table 1).

Ring Ouzels were detected between April and July although sites were not visited before or after those dates (Table 2). Males used boulders or rock outcrops on prominent ridges as song perches. Males sang in the early morning from at least 06.10 hrs (time of first arrival on transect route). Birds were most vocal in the early morning (06.10-09.30hrs) with only very intermittent song after 09.30-10.00 hrs. No birds were detected by song after 11.45 hrs. Although few sites were visited systematically in the pre-dusk period, Ring Ouzels were recorded singing at three sites up to dark (20.00-21.30 hrs).

Singing males and/or nesting pairs were located between 400-850 m a.s.l. Mean elevation of sites did not appear to vary significantly over the study period although the elevational range of site contracted with sites at low and high elevation becoming unoccupied (Table 2). Likewise, the apparent 'density' of Ring Ouzel as measured by 'nearest neighbour

Table 1. Ring Ouzel occupancy at sites in the Reeks in 2008-2011, and 2012-2017 (not all sites visited in each year). Shaded sites were not visited in that year. Note that O = Occupied, X = not occupied.

Year	2008	2009	2010	2011	2012-2017
Reeks (core sites)					
1	O	X	X	X	X
2	O	O	X	X	X
3	O	O	O	X	X
4	O	X	X	X	X
5	O	O	X	X	X
6	O	X	X	X	X
7	O				
8	X	O	X	O	X
9	O	X	O	O	O
10	O	O	O	O	O
11	O	O	O	O	O
12	O	O	X	X	X
Ex Reeks sites					
Mangerton area		X		X	X
Other sites*		X	X	X	X
Sites occupied (breeding pair estimate)	**11** (6-10)	**6/7** (5-7)	**4** (4-6)	**4** (3-5)	**3** (3) (minimum)

*Sites (n=3) on the Iveragh peninsula known to have been occupied in the past (Carruthers 1998).

Table 2. Behavioural and ecological data for Ring Ouzels in the Reeks in 2008-2011. Elevation was based on singing males or nests. Nearest neighbour distance (NND) was the mean distance between neighbouring males/nests.

Year	Sites checked	Sites occupied	No. of visits	Survey period (dd/m)	Song detection range (hrs)	Elevation range (m) $\bar{x}$ = mean	NND (km)
2008	12	11	1	02/5-27/5	0700-1058	450-820 ($\bar{x}$ = 657)	2.03
2009	15	6/7	2	21/4-03/6	0640-1145	400-800 ($\bar{x}$ = 624)	2.37
2010	17	4	1-4	21/4-23/7	0700-1130	400-850 ($\bar{x}$ = 650)	1.50
2011	14	3+	1-2	29/4-18/5	0610-1115	500-850 ($\bar{x}$ = 675)	2.24

Plate 7. Ring Ouzel breeding habitat in the MacGillycuddy's Reeks, Co. Kerry (A. Mee).

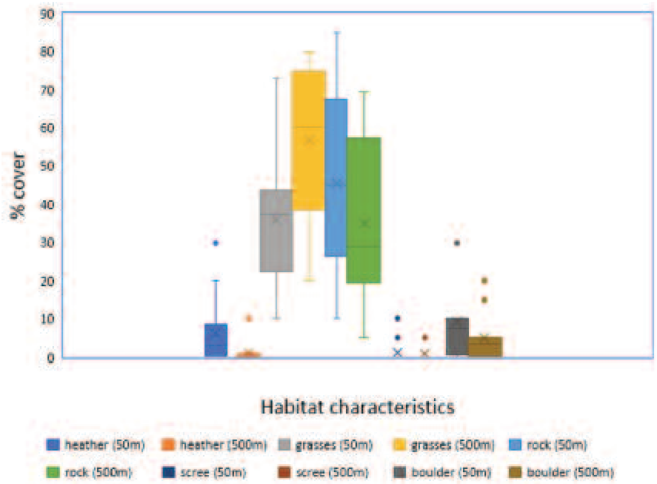

Figure 3. Habitat characteristic of Ring Ouzel sites at 50m and 500m radii of nests and/or singing males. Boxplots show the 50% quantiles (box), median (horizontal bar), range and outliers (dots). There were too few values for scree (values shown as dots).

distance' (the distance between neighbouring singing males or nest sites) varied little between years except in 2010 when one site (site 8) was unoccupied (Table 2). Instead there was an apparent consolidation of sites with the loss of birds at 'peripheral' sites with the population contracting to three 'core' sites, i.e. sites that remained occupied in most or all years.

Even though Ring Ouzels are highly territorial, interactions between neighbouring birds were rarely recorded. However, a male observed singing and preening over a 10-15 minute period, from the usual perch of the 'resident' male above a nest site (site 10) in June 2010, was subsequently chased off by the presumed resident male to more than 1km east towards the neighbouring territory (site 9) before returning a short time later. Some males at nest sites on the north side of the main ridge of the Reeks were observed to apparently cross the highest ridges to sing on the south side of the ridge although usually within 0.5km of nest sites. Males were seen with females during periods off nests, apparently acting as 'lookouts' while the female actively foraged rather than actively engaging in foraging themselves. Such close 'guarding' could be interpreted as anti-predator vigilance or possibly mate guarding early in the breeding attempt where intruding males may attempt to copulate with the female.

Habitat

Ring Ouzels were mainly observed on very steep cliffs and ridges in largely grass dominated habitats with a high proportion of exposed rock (Figure 3). The few nest sites located were on largely inaccessible steep-vertical cliffs with heather and Greater Woodrush *Luzula sylvatica* on ledges and crevices. Cliff nest sites appeared to support a greater vegetation cover (heather, woodrush) than apparently suitable but unoccupied cliffs nearby.

Of the 12 sites holding singing or breeding Ring Ouzel only three had 10% or more heather cover (0-30%) at the 50 m scale but only one (site 1) had >5% heather cover (10%) at the 500 m scale (Figure 3). All territories had extensive grass cover (30-80%) and exposed rock (15-70%) at 500 m, most with boulders on the lower slopes. Only three sites held any scree cover although extensive scree slopes exist in many parts of the Reeks. All were located on very steep-vertical slopes 150-580 m from open water (lakes or streams), although several sites held less discernible or ephemeral water sources (e. g. non-calcareous springs, temporary streams after heavy rain).

Habitat use by Ring Ouzels away from prominent song perches or nest sites was difficult to quantify as prolonged observations of birds were rare except for singing males. However, at one site both male and female were observed flying out from the nest to feed in moss-dominated non-calcareous springs (wet flushes). Birds turned over clumps of moss when foraging, presumably to find invertebrate prey. At other sites, females apparently left nests to forage on occasion in nearby flushes before returning to incubate or brood young. Foraging females away from nests seemed to favour such wet flushes, comprising moss covered areas at the base of steep slopes within 0.5 km of nest sites.

Plate 8. Ring Ouzel breeding habitat in the MacGillycuddy's Reeks, Co. Kerry. The foreground shows the contrast between the lush vegetation cover on the lake island and the steep slopes in the background (A. Mee).

Discussion

The MacGillycuddy's Reeks held a small population of Ring Ouzel during the main study period 2008-2011. However, the most easily accessible and regularly occupied sites (10, 11) were checked every year from 2008 to 2017 and held potential breeding Ring Ouzels (singing males or pairs) in all years. The population of Ring Ouzels monitored in the Reeks appeared to decline during the period of study, from a high of 11 apparently occupied sites in 2008 to just four by 2011. However, only one visit was made to all sites in 2008. Ring Ouzels are known to sing on migration (Cramp & Simmons 1998). Thus, some of the birds detected may have been migrants *en route* to breeding sites further north, in the UK or Scandanavia. At least two visits are required to accurately determine occupancy and thus the size of the breeding population (Gilbert *et al.* 1998). Further, while only three sites were occupied between 2012 and 2017, coverage was not as comprehensive in those years and not all sites were surveyed in all years (Table 1).

Despite these caveats it is evident that the observed Ring Ouzel population in the Reeks has declined in recent years. Only two sites (sites 10 & 11 in Figure 2, Table 1) have been occupied consistently over the study period and along with a third site (site 9), appear to form the 'core' of the small remnant population. All three sites are immediately adjacent to each other and located in north facing coums. This may be important in itself as higher summer temperatures resulting from climate change have been suggested as one of the drivers of population decline in Britain (Beale *et al.* 2006). One possibility is that warming results in drier soils and reduced earthworm availability (Sim *et al.* 2010). If this is true, then ouzels nesting in the locally cooler and more moist conditions in north-facing sites may be better able to persist at least in the short-term. However, an intensive check of this area (sites 10, 11) on one date in 2018 failed to locate any singing birds (Alan McCarthy pers. comm.). Regardless, the population is very small and vulnerable to local extinction (Purvis *et al.* 2000).

It is likely that the Reeks population has been in decline for some considerable time, in tandem with the significant declines in the species range in Ireland during the 20th century (Ussher & Warren 1900, Kennedy *et al.* 1954, Balmer *et al.* 2013). The Ring Ouzel range declined by 57% in Ireland in the 40 years between the first *Breeding Atlas* in 1968-72 and the most recent in 2007-11 (Balmer *et al.* 2013). However, there were no confirmed or probable breeding records for Ring Ouzel in the Reeks in the 1968-72 *Breeding Atlas* (Sharrock 1976). In contrast to continued declines elsewhere in Ireland, the second *Breeding Atlas* in 1988-91 (Gibbons *et al.* 1993) found that "there has been an interesting colonisation of Co. Kerry." However, a much more likely explanation for this apparent increase is that much or all of the breeding population went undetected in the 1968-72 survey (Hutchinson 1989). Much of the terrain occupied by Ring Ouzels in the Reeks is difficult to access and, as this study suggests, survey work after 09.30hrs would have likely missed most singing males. Moreover, the second *Breeding Atlas* coincided with the first serious survey of the species in Kerry (Carruthers 1998). Carruthers (1998) estimated the breeding population in Kerry to be probably no more than 10 to 15 pairs, located mainly in the Reeks, but also the Mangerton area and at a site on the Dingle peninsula. Evidently Ring Ouzels have further declined in both range and population size since the early 1990s. This study failed to locate birds in the Mangerton area despite the apparent suitability of much of the habitat in that area. Likewise, there have been no recent records of Ring Ouzel in the breeding season in the Dingle peninsula despite some dedicated efforts to locate singing males (Michael O'Clery pers. comm.). Thus, it appears likely that the population has contracted to a tiny core area in the central Reeks.

Recent reports of breeding Ring Ouzel elsewhere in Ireland have been scarce in the last 10-20 years (see Annual Reports of Irish Rare Breeding Birds Panel). However, as in Kerry, a small population continues to persist in south-west Donegal (Cox *et al.* 2002, McGeehan & Wyllie 2012, Balmer *et al.* 2013). Outside of Kerry and Donegal reports of ouzels in breeding habitat in summer are very scarce (Perry & Newton 2014). A pair probably bred at a site in Sligo in 2015 (see Newton 2016) and single birds have been recorded in the Wicklow and Tipperary, although without breeding evidence (eg., see Hillis 2008).

Much of the Reeks is devoid of heather cover apart from the western slopes and a few outlying hills, the Gap of Dunloe and the Tomies-Purple Mountain area. A mosaic of heather patches as cover and open areas to forage appear to be important habitat preferences for Ring Ouzels in Britain and this is also likely to be the case in Ireland. Sim *et al.* (2007) found that ouzel breeding sites in south-east Scotland were composed of heather or grass-heather mosaics within 100m of nests. Moreover, those territories at higher elevation and with greater heather cover were more likely to persist as breeding sites compared to those that became defunct. Likewise, the abundance of Ring Ouzels on a national scale in Scotland was also associated with heather-grass mosaics (Buchanan *et al.* 2003). Cover provided by heather in the vicinity of nests appears to be a key criterion in nest site selection while short grass for foraging nearby appears to be important during the nestling phase (Sim *et al.* 2007). Cover provided by vegetation, especially heather, may also provide protection from potential predators. Anecdotal evidence from observation of Ring Ouzels in song in the Reeks during this study suggest that birds may respond to the presence of potential nest predators such as Common Ravens *Corvus corax* by becoming quiet. Ravens were sometimes attracted into ouzel breeding areas in the Reeks by the presence of sheep carrion, possibly due to falls from steep cliffs. Avian predators can be an important factor in Ring Ouzel breeding success and post-fledging juvenile survival (Smith 2006, Sim *et al.* 2013). This is also likely to be the case in Ireland where the loss of heather cover on many former Ring Ouzel breeding haunts is likely to increase predation risk for fledged young (Sim *et al.* 2013).

The increasing human footprint as a result of steadily increasing numbers of recreational users in the Reeks (125,000 reported accessing the Reeks in 2017, MacGillycuddy's Reeks Mountain Access Forum) may also impact Ring Ouzels by increasing disturbance at sensitive sites. Nest desertion and failure due to disturbance from walkers and rock climbers has been an important factor in the Peak District, Britain (Melling 2003, Leyland 2016). Additionally, high densities of recreational users and their subsequent food or waste remains may also increase predation pressure by attracting corvids to remote areas. Although most of the known Ring Ouzel sites in the Reeks are unlikely to be directly impacted by disturbance due to their location away from the most popular hiking routes, one of the 'core' sites (site 11) is located within metres of an increasingly busy climbing and walking route. Further, Ravens are a constant presence in the Corrán Tuathail area possibly due to the presence of humans and the associated food waste discarded by hikers.

Nest sites favoured by the remnant Irish population appear to be similar to British populations with the few nest sites located in the Reeks being characterised by some heather and woodrush cover. Heather cover also appears to characterise some of the remaining Ring Ouzel sites elsewhere in Ireland such as in Donegal (Cox *et al.* 2002). It is likely that decades, if not centuries of intensive grazing, principally by sheep, and burning has had a deleterious effect on the extent of heather cover in the Reeks. Loss of heather cover resulting from burning followed by grazing has well documented in other upland mountain areas in recent decades, such as the Galtee Mountains (Tipperary-Limerick), where year-round sheep grazing has replaced the booley system (transhumance) of summer grazing by cattle in the uplands, practiced in the Galtee Mountains up until the latter half of the 19th century (Costello 2016).

Ring Ouzel declines in Scotland have also resulted from habitat loss to large-scale afforestation with the negative effects of afforestation also appearing to extend outside the 'forest footprint' perhaps due to habitat fragmentation and increased predation risk (Buchanan *et al.* 2003). Large-scale afforestation commenced in Ireland after declines in Ring Ouzel populations were underway (Kennedy *et al.* 1954) and little afforestation has taken place in the Reeks to date, indicating that afforestation is unlikely to explain declines in the Reeks. However, it is plausible that large scale afforestation in former strongholds such as the Wicklow mountains (see Sharrock 1976) may have accelerated such declines directly due to habitat loss and indirectly due to increased predation risk in the remaining unplanted habitat fragments.

A perhaps neglected area of research has been post-breeding dispersal in the breeding areas (Sim *et al.* 2013). Ring Ouzels are especially dependent on crops of berries of various plants in the post-fledging dispersal period, most notably Bilberry, Crowberry and Rowan (Burfield 2002, del Hoyo *et al.* 2005). Crowberry is scarce in Ireland except in the North-west where it is at the southern/western edge of its European range, and so is unlikely to have been an important food source for Ring Ouzels in Kerry. Thompson (1849) noted that Ring Ouzels are "stated to appear there (mountains of Dublin and Wicklow) in flocks in spring and autumn, at the latter season to eat the berries of the mountain ash." Rowan occurs sparsely in the Reeks as does bilberry, probably due to long decades of browsing by livestock, mainly sheep and goats (Hester *et al.* 1998). The latter may be important in that goats can access cliffs and other precipitous slopes that hold remnant rowan. However, there is some anecdotal evidence in the Reeks of small groups of Ring Ouzel, possibly post-fledging family parties, being encountered on the slopes of Mangerton, where there is relatively good growth of berry-bearing shrubs, especially Bilberry. It is likely that family parties of adults and juveniles may disperse locally in late summer to areas with good berry crops. The importance of these 'post-fledging dispersal' areas for juvenile survival may be critical but we know little about the extent and continued use, if any, of these areas.

While we know little of the history of grazing in the Reeks, Weld (1807) gives some insight into this on an expedition to Corrán Tuathail in the early 19th century: "On the summit of this mountain (Strickeen Mountain) we found an extensive tract of ground, less encumbered by rocks than the valley below, and covered as far as the eye could see with heath and coarse grass, on which innumerable herds of cattle were fed." It is possible that a shift from summer grazing on the slopes of the Reeks to year-round sheep grazing has had a detrimental effect on the vegetation, gradually replacing upland heath with grasses. Studies have shown that excluding sheep from heavily degraded and species poor upland blanket bog can have restorative effects including crowberry and heather regeneration (Rawes 1983) but this effect may be negated by increased deer grazing and/or burning (Hope *et al.* 1996). However, ceasing grazing in large-scale grass dominated uplands such as the Reeks is likely to be impractical for social reasons where local communities are largely dependent on extensive sheep grazing. Moreover, the very low base of heather cover is likely to preclude successful regeneration. Experimental light mixed grazing allied to seeding has been shown to be successful (Mitchell *et al.* 2013) and could be trialled in the Reeks. Furthermore, observations, albeit limited, suggest that non-calcareous springs (wet flushes) may be important for foraging Ring Ouzels during the breeding season. Buchanan *et al.* (2006) note that management which creates a mosaic of habitats and the presence of wet flushes associated with spring emergence of adult insects may increase invertebrate food for birds.

Although the global conservation status of Ring Ouzel is classed as of least concern (del Hoyo *et al.* 2005), populations within Ireland and the UK are in serious decline. Thus, the species is red-listed and at risk of extinction in Ireland (Colhoun & Cummins 2013). Habitat change in the uplands, loss of heather cover, low survival rates, effects on wintering grounds, hunting pressure on migration, predation, and climate change have been cited as driving declines in this enigmatic upland bird species (Burfield 2002, Buchanan *et al.* 2003, Beale *et al.* 2006, Ryall & Briggs 2006, Sim *et al.* 2010, 2011, 2013, Wotton *et al.* 2002). Ring Ouzels lack the protection provided by conservation measures put in place under the EU Birds Directive (2009/147/EC) such as the designation of Special Protection Areas as they are not an Annex 1 species. Moreover, a Species Action Plan has yet to be developed although a number of important targetted actions have been proposed for key upland bird species including Ring Ouzel (BirdWatch Ireland 2010) but these remain to be implemented. Areas of critical importance for the few remaining remnant populations within Ireland should be identified and given high conservation priority. A comprehensive survey of the remnant populations, where they persist, in the Reeks and Donegal is urgently needed. Habitat management to increase heather cover and enhance or maintain suitable heather-grass mosaics at key sites should be an important and urgent conservation measure. This could be the basis of local results-based agri-environment projects or schemes targetted in the immediate term on the few remaining key sites. Further research including determining factors influencing nest success and post-fledging survival rates, identifying important post-fledging foraging areas are also important in understanding the factors underlying population trends.

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Barnacle Geese *Branta leucopsis* in Ireland: results of the 2018 census

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A complete aerial and ground census of Barnacle Geese *Branta leucopsis* was conducted in Ireland in spring 2018 as part of the international Greenland Barnacle Goose census. This population winters exclusively in north-western Ireland and Scotland, where regular censusing is ongoing since 1959. A total of 16,237 Barnacle Geese was recorded in Ireland in March 2018, primarily on the north-west coast and offshore islands. The abundance of Barnacle Geese in Ireland has been on a general upward trend since the 1950s. However, the 2018 census represents a decrease of 7% compared to the 2013 estimate, in accordance with a flyway population decline in recent years. Internationally significant flocks were recorded at Ballintemple, Dunfanaghy New Lake, Trawbreaga Bay, the Inishkea Islands, Cross Lough and Termoncarragh. A further 11 sites held nationally important numbers and a high proportion of the population was associated with the European Union Birds Directive Special Protection Area network. There was no notable reduction in the range of this species in Ireland, nor a reduction in the proportion of the flyway population wintering here when compared with the last census in 2013. The five-yearly census continues to provide useful data for long-term monitoring.

Introduction

The Greenland Barnacle Goose *Branta leucopsis* winters exclusively in north-western Ireland and Scotland. The entire population migrates to north-east Greenland for the breeding season, staging in north-west Iceland on the spring journey and in south-east Iceland in autumn. In the 1950s, this population was considered threatened due to its declining trend. Protective legislation was introduced, and several winter censuses were conducted to monitor the recovery (Boyd 1968). A regular international census of Greenland Barnacle Geese in Ireland and Scotland began in 1959 and continues to the present day (Boyd 1961, Mitchell & Hall 2013).

Barnacle Geese in Ireland principally occur on offshore islands and along the coasts of counties Donegal, Sligo, Mayo and Galway (Crowe *et al.* 2014). Smaller numbers can be found in counties Clare (e.g. Mutton Island), Kerry (e.g. the Magharee Islands) and Wexford (e.g. the Slobs) (Merne &

Plate 9. Barnacle Goose at Ballyconnell in County Sligo (Ulrike Schwier).

Walsh 2003, Crowe *et al.* 2014). The international censuses have shown that the Irish portion of the flyway population has increased several-fold since the 1950s: from 2,800 individuals in 1959 to 17,500 in 2013 (Boyd 1961, Crowe *et al.* 2014).

A complete aerial and ground census of Barnacle Geese was conducted in Ireland in spring 2018 as part of the international Greenland Barnacle Goose census. Here we present results of the 2018 census and discuss historical trends and current status of the species in Ireland.

Methods

Previous censuses

Full censuses of Barnacle Geese in Ireland were conducted 13 times between 1959 and 2013. Following an initial intense survey effort, censuses were conducted at approximately five-year intervals. The dates of the initial censuses were March and December 1959 and spring 1961, 1962, 1965, 1966 and 1973 (Boyd 1968, Ogilvie & Boyd 1975). Periodic spring censuses were conducted from 1975 on (Ogilvie & Boyd 1975, Ogilvie 1983, Walsh & Merne 1988, Merne & Walsh 1994, 2002, Walsh & Crowe 2008, Crowe *et al.* 2014).

2018 census

A full aerial and ground census of known Barnacle Goose sites in Ireland was undertaken from 19 to 21 March 2018. Ground counts took place on 19, 20 and 21 March. Sites covered during the ground-based census were: Dooey, Inishowen, Dunfanaghy (Donegal), Ballintemple (Sligo), Achill, Bellmullet, Clew Bay (Mayo) and the west Clare coast.

The aerial census took place on 19 and 20 March using a Cessna 172. On the first day, the aircraft departed from Weston Airfield at 11.00 hours GMT and the survey transect commenced at the Blasket Islands (Kerry), moving northwards to Blacksod Bay (Mayo). The survey transect was completed at Strandhill (Sligo) at 18.15 hours. On the second day, the aircraft departed Strandhill at 08.30 hours and the survey transect continued northwards to Inishtrahull Island (Donegal), finishing at 10.35 hours. Survey methodology followed that outlined in Walsh and Merne (1988). Observers at either side of the aircraft made counts of flocks flushed from islands and coastal sites during the transect. Any large flocks were photographed. Following the survey, flocks in photographs were counted by two independent counters to ensure accuracy, and these were collated with ground and aerial counts to provide a final total number. Where multiple estimates existed for the same flock, the photograph was taken as the best estimate, followed by the aerial count and lastly the ground count.

Results

2018 census

In 2018, a total of 16,237 Barnacle Geese was recorded in Ireland. A total of 58 sites was visited (Appendix 1). Of these, 33 sites supported Barnacle Geese (Figure 1). The largest flocks were recorded at Ballintemple (Sligo), the Inishkea Islands (Mayo), Trawbreaga Bay and Dunfanaghy New Lake (Donegal). Ballintemple supported the highest number of geese, at 4,410, followed by the Inishkea Islands, at 2,330.

Figure 1. Sites supporting Barnacle Geese during the March 2018 census. Increasing abundance is represented by increasing circle size, from the smallest flocks (5–250 individuals), to small (251–800 individuals), mid-range (801–1,500 individuals), large (1,501–2,500 individuals), through to the largest flocks (2,501–4,500 individuals).

There were nationally (≥ 160 individuals) and internationally (≥ 720 individuals) important numbers of Barnacle Geese at several sites (Table 1). Six sites supported internationally important numbers (1% of flyway population). These were Ballintemple (Sligo), Dunfanaghy New Lake and Trawbreaga Bay (Donegal), as well as the Inishkea Islands and nearby mainland sites Cross Lough and Termoncarragh on the Mullet Peninsula (Mayo). A further 11 sites held nationally important numbers (1% of national population). These were

Table 1. Sites where numbers of Barnacle Geese exceeded internationally and nationally important thresholds during the March 2018 census.

County	Site	Total Count
Sites exceeding international threshold		
Mayo	Inishkea Islands* (F555214)	2,330
Mayo	Cross Lough* (F639294)	804
Mayo	Termoncarragh* (F650349)	940
Sligo	Ballintemple* (G644436)	4,410
Donegal	Dunfanaghy New Lake* (C000363)	1,300
Donegal	Trawbreaga Bay* (C436514)	1,775
Sites exceeding national threshold		
Galway	Birmore Island* (L801262)	587
Galway	St. MacDara's Island* (L721299)	221
Galway	Croaghnakeela Island* (L687323)	252
Galway	Inishshark Island* (L484648)	638
Mayo	Moynish More Island (L862943)	169
Mayo	Tiraun (F617237)	184
Mayo	Annagh Head (F639341)	243
Donegal	Dooey (C088421)	450
Donegal	Inishdooey Island* (B896383)	280
Donegal	Doagh* (C086417)	300
Donegal	Malin Head (C402592)	380

Flocks associated with Special Protection Areas are marked with an asterisk. International threshold is 720 and national threshold is 160, based on the 2018 census.

Birmore Island, St. MacDara's Island, Croaghnakeela Island, Inishshark Island (Galway), Moynish More Island, Tiraun, Annagh Head (Mayo), Dooey, Inishdooey Island, Doagh and Malin Head (Donegal). Of all birds recorded, 89% were associated with Special Protection Areas (SPAs) designated under the European Union Birds Directive (Directive 2009/147/EC) (Table 2). Barnacle Geese are a listed Special Conservation Interest within all SPAs in which they were recorded, with the exception of Inishmore, Mullet Peninsula, Blacksod Bay/Broad Haven and West Donegal Coast SPAs.

Table 2. County totals for the number of Barnacle Geese recorded during the March 2018 census. The proportion of birds recorded on offshore islands and associated with Special Protection Areas (SPAs) is also indicated, along with the number of sites Barnacle Geese were recorded at.

County	Number of birds	% on islands	% in SPAs	Number of sites
Clare	30	100%	100%	1
Galway	1,948	100%	59%	7
Mayo	4,984	51%	15%	12
Sligo	4,410	0%	100%	1
Donegal	4,865	12%	18%	12
Overall	**16,237**	**32%**	**89%**	**33**

Population trends in Ireland

The 2018 census estimate of 16,237 birds represents a decrease of 7% compared to 2013 (Figure 2). The proportion of the flyway population wintering in Ireland in 2018 was 23% (Figure 2). This is very similar to the 2013 figure of 22%. In accordance with the overall population decline, the number of birds in most counties also decreased between 2013 and 2018, with the exception of Galway (Table 2). No Barnacle Geese were recorded in Wexford or Kerry during this census.

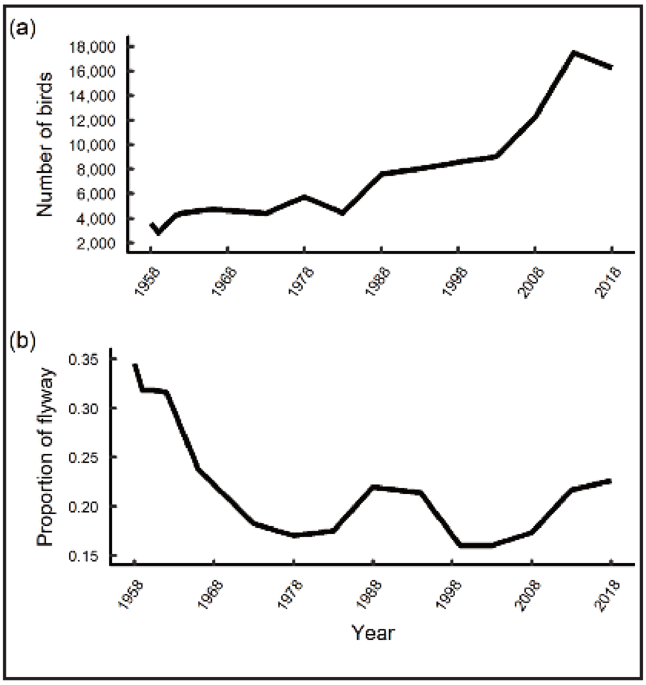

Figure 2. Population trend of Barnacle Geese in Ireland between 1958 and 2018 (a) and trends in the proportion of the flyway population in Ireland between 1958 and 2018 (b) (source of 1958 to 2013 data: Crowe *et al.* (2014)).

Discussion

The 2018 census of Greenland Barnacle Geese in Ireland was part of the most recent international survey of the flyway population. Successful coverage of known Barnacle Goose sites in Ireland was achieved, with good survey conditions for both ground counts and the aerial transects. A total of 16,237 Barnacle Geese was recorded in Ireland in March 2018, primarily on the north-west coast and offshore islands.

Long-term trends

The abundance of Barnacle Geese in Ireland has been on a general upward trend since the 1950s, in line with many northern migrant goose species (Fox & Leafloor 2018). The considerably depleted population of 2,800 birds in 1959 increased and remained relatively stable around a figure of 5,000 individuals for two decades, aside from a brief decline in the late 1970s (Ogilvie & Boyd 1975, Ogilvie 1983). From the early 1980s, the population grew rapidly, increasing four-fold to a high of 17,500 in 2013 (Crowe *et al.* 2014). The global increase in the Barnacle Goose and other goose species has been attributed to improved food resources as geese transition from traditional to agricultural habitat (Fox & Abraham 2017, Clausen *et al.* 2018). It has also been demonstrated that warmer and wetter winter conditions as a result of climate change, have improved survival and productivity prospects in waterfowl (Kéry *et al.* 2006, Dickey *et al.* 2008, Cleasby *et al.* 2017, Guéry *et al.* 2017).

The 7% decrease in Ireland's Barnacle Goose numbers observed over the most recent five-year census interval is the first since the late 1970s, although it is relatively small. The cause of the 23% decrease observed during the 1978 to 1983 census interval was unclear. Ogilvie (1983) suggested the prolonged drought in summer 1976 resulted in little food for geese the following season and this, along with a very cold winter in 1978-79, may have had a negative impact on the population through food shortage. Geese are highly susceptible to unanticipated changes in otherwise predictable food resources (Clausen *et al.* 2012).

The remarkable rate of increase in Ireland's Barnacle Geese observed since 2001 has clearly reduced. The population size increased by 43% in the 2008 to 2013 census interval and by 35% in the preceding 2003 to 2008 interval. It is unclear whether the recent decrease represents a period of population decline or a plateau in the period of rapid increase. We suspect there has been a decrease in Barnacle Goose immigration into Ireland due to high levels of shooting mortality on the Isle of Islay in Scotland (summary of bag data available from Scottish Natural Heritage: https://www.nature.scot/goose-management-scheme-islay-documents). Future surveys should provide greater insights. Nevertheless, the population in 2018 is still several times higher than the critically low numbers reported during the 1950s, and it currently appears to be secure. There has been no notable reduction in the range of this species in Ireland, nor a reduction in the proportion of the flyway population wintering here when compared with 2013.

The population decrease in Ireland mirrors the trend in Scotland. In Scotland, the population decreased by 12% in the 2013 to 2018 census interval (C. Mitchell, unpublished data). That both national trends are falling indicates there is a decline in the flyway population rather than geese simply shifting their wintering distribution from Ireland to Scotland. Barnacle Goose flocks in Ireland exceeded the international threshold in similar geographical areas to those recorded in the 2013 census (Crowe *et al.* 2014). At a meta-scale, there is consistency in the location of internationally significant flocks over time: internationally significant numbers are repeatedly recorded around the Mullet Peninsula, Ballintemple, Dunfanaghy New Lake and the Inishowen Peninsula.

Plate 10. Barnacle Geese (www.carlmorrowphotography.com)

Distribution

Barnacle Geese in Ireland in 2018 were highly concentrated in the north-west. A small proportion of the population was recorded in Clare, but the majority was found from coastal counties between Galway and Donegal. Mayo supported the greatest abundance, closely followed by Donegal and Sligo (where the total comprised just a single flock of over 4,000 birds in Ballintemple). Roughly a third of the population was recorded from offshore islands (Table 2), the most noteworthy in terms of numbers being the Inishkea Islands, supporting an internationally important population. The remaining two thirds of the population was recorded from coastal mainland areas, with the largest flocks at Ballintemple, Dunfanaghy New Lake and Trawbreaga Bay. Donegal had the greatest distribution of geese, as flocks were recorded at 22 locations, perhaps reflecting smaller areas of suitable habitat available.

Given the high mobility of these geese, it is important to consider seasonal movements of the population – for example, a move northward prior to spring migration (Philips *et al.* 2003) – when analysing patterns of occurrence. This census has traditionally been carried out during early spring and may not accurately reflect the distribution of birds during the mid-winter period. The survey timing may explain the absence of geese in Kerry (the southernmost part of the range) and the small numbers in Clare. Wintering range retraction or short-stopping is an unlikely cause as annual Irish Wetland Bird Survey (IWeBS) data shows no decreasing trend in the numbers of Barnacle Geese in the south-west (Tralee Bay and the Clare coast) during mid-winter (IWeBS Office, unpublished data). Small flocks (up to 32 individuals) were recorded at the Slobs in Wexford up to March 2016 (IWeBS Office, unpublished data), while a flock of 14 was recorded in the Wexford Wildfowl Reserve in February 2018 (A. Walsh, pers. obs.).

The 2018 population is almost twice its 2003 size (Merne & Walsh 2003). The period from 2003 to the present saw a great increase in the abundance of geese at mainland sites when compared with islands, particularly in Donegal and Mayo (Merne & Walsh 2003). If this is a general pattern of change which continues, it has implications for the agricultural community as Barnacle Geese tend to feed on agricultural grassland, sometimes with considerable negative impacts (Bainbridge 2017, Mason *et al.* 2017). A high proportion (>80%) of birds on the mainland were associated with the SPA network. Agri-environment schemes, both within and outside of the SPA network, have the potential to be instrumental in the conservation management of this species into the future. Numbers outside the SPA network were highest on the Mullet Peninsula, Dooey and Malin Head, thus could represent a future management challenge. Units of management of Greenland Barnacle Geese outside the breeding grounds at the Irish and Scottish scale also needs to be further explored, as conservation management strategies for one country is likely to influence the other. For example, ongoing derogation shooting on Islay (McKenzie 2014) is likely to impact Irish totals as there is high connectivity between sites. Even localised disturbance of geese can result in impacts at the flyway level (Klaassen et al. 2006, Jensen *et al.* 2017).

Future monitoring

The five-yearly international census of Greenland Barnacle Geese continues to be a necessary tool in long-term monitoring of the population. Robust estimates of the population can only be captured in these thorough surveys, along with national scale changes in distribution. Such data provides the evidence base to inform the conservation management of the 23% of the flyway population of Greenland Barnacle Geese that overwinter in Ireland. Full national surveys during the census intervals would also be useful in determining annual variation and site use, particularly during the autumn and mid-winter periods. This would be valuable in targeting management actions appropriately to mitigate potential future challenges.

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Appendix 1.

Sites visited during the March 2018 Barnacle Goose census.

Kerry
Beginish Island (V410787)
Magharee Islands (Q621213)
Clare
Mutton Island (Q971744) 30*
Galway
Aran Islands (L901062) 20
Birmore Island (L801262) 587*
St. MacDara's Island (L721299) 221*
Croaghnakeela Island (L687323) 252*
Inishturk Island (L595746)
Cruagh Island (L530550)
High Island (L503574)
Aughrus Point (L544572)
Friar Island (L523578)
Inishshark Island (L484648) 638*
Crump Island (L676656) 115
Davillaun Island (L582663) 115*
Mayo
Emlagh Point (L741797)
Caher Island (L660760)
Roonagh Lough (L747766)
Clare Island (L680850)
Achill Beg Island (L710923)
Moynish More Island (L862943) 169
Inishgallon Island (F622030)
Duvillaun Island (F572159) 60*
Falmore (Mullet) (F627185) 81
Surgeview (Mullet) (F609188) 17
Eachléim (Mullet) (F620208) 142
Inishkea Islands (F555214) 2330*
Tiraun (Mullet) (F617237) 184
Elly (Mullet) (F629244) 8
Barnagh (Mullet) (F653268) 6
Cross Lough (Mullet) (F639294) 804
Inishglora Island (F611311)
Annagh Head (F639341) 243
Termoncarragh (F650349) 940*
Sligo
Ballintemple (G644436) 4410*
Innismurray Island (G571540)
Donegal
St. John's Point (G704695)
Shalwy (G639739) 60
Inishduff Island (G647723)
Muckros Head (G622737)
Fintragh Bay (G678761)
Rathlin O'Birne Island (G466801) 110*
Inishkeel Island (B704000) 6*
Dooey (C088421) 450
Roaninish Island (B656026)
Inishkeeragh Island (B683122) 133*
Aranmore Island (B663157) 11
Owey Island (B711231)
Inishfree Lower Island (B756240)
Falcarragh (B934339)
Dunfanaghy New Lake (C000363) 1300
Inishdooey Island (B896383) 280
Inishbeg Island (B895396) 60
Doagh (C086417) 300*
Fanad Head (C227477)
Trawbreaga Bay (C436514) 1775*
Malin Head (C402592) 380*
Inishtrahull Island (C480654)

An asterisk indicates Special Protection Area. A count is given where Barnacle Geese were recorded.

Diet of the Barn Owl *Tyto alba* in east County Cork

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Keywords: Barn Owl, *Tyto alba,* diet, invasive mammals

Barn Owls *Tyto alba* in east County Cork fed on all the small mammal species available to them, including the invasive Bank Vole *Myodes glareolus*. Commonest in the diet was the Field Mouse *Apodemus sylvaticus* and Bank Vole (by number and biomass). The House Mouse *Mus domesticus* and the Pygmy Shrew *Sorex minutus* were next in importance (by number) although the Pygmy Shrew was insignificant in biomass due to its small size. The Brown Rat *Rattus norvegicus* was the third most important species in biomass but was only fifth in importance by number. Bats, birds and the Common Frog *Rana temporaria* each formed a low proportion of the diet. A comparison of data collected at the same roost sites within three District Electoral Divisions (DEDs) in 1992-94 with data from the present study collected in 2003-06 and 2011-16 shows that the Bank Vole, since its colonisation of the area, accounts for almost 30% of prey by numbers and biomass.

Plate 11. Barn Owl (Michael O'Clery).

Introduction

The feeding ecology of the Barn Owl *Tyto alba* has been studied in some detail across its geographical range in North and South America, Europe, Asia, Africa and Australia (Bunn *et al.* 1982). The feeding ecology of this owl has been studied in Ireland only since the 1960s (Smiddy *et al.* 2018), although the subject is of considerable interest for several reasons (a) the low number of indigenous mammals of suitable size available as prey, and (b) the recent arrival (probably through accidental introduction) of two prey species. Traditionally, Barn Owls in Ireland had the Brown Rat *Rattus norvegicus,* the Field Mouse *Apodemus sylvaticus,* the House Mouse *Mus domesticus* and the Pygmy Shrew *Sorex minutus* available to them as mammal prey. These mammals are widespread across the island (Lysaght & Marnell 2016), and can occur in a variety of habitats with heavy ground cover, while two of them (Brown Rat and House Mouse) are commensals of man and can be found in and around farmyards, and in urban and industrial landscapes.

The two recent arrivals are Bank Vole *Myodes glareolus* (Smiddy 2016) and Greater White-toothed Shrew *Crocidura russula* (McDevitt 2016). The Bank Vole is likely to have been present in Ireland for about 100 years (Stuart *et al.* 2007), while the Greater White-toothed Shrew is present for about 20 years (Tosh *et al.* 2008). Both have restricted ranges; the Bank Vole being present over about one-third of the island in the south and west (Smiddy 2016), while the Greater White-toothed Shrew is confined to a much smaller area in a few southern counties, largely within the range of the Bank Vole (McDevitt *et al.* 2014, McDevitt 2016).

The diet of the Barn Owl has been studied inside and outside the range of the Bank Vole in Ireland. Most studies have been small and restricted in geographical terms and typically involved collections of pellets which yielded less than 1,000 vertebrate prey items, with only four studies exceeding 5,000 vertebrate prey items (Smiddy *et al.* 2018). The largest study for which data has been published to date assessed diet in County Cork and involved almost 11,000 vertebrate prey items (Cooke *et al.* 1996). Most data for that study were collected in 1992-94 and involved Barn Owl pellets collected from all parts of the county. The present paper reports on collections of Barn Owl pellets made in east County Cork during 2003-06 and 2011-16 at three District Electoral Divisions (hereafter DEDs) where collections were also made in 1992-94.

Methods and study area

The diet of the Barn Owl was studied in County Cork in 1992-94 (Cooke *et al.* 1996) and data presented for different DEDs. For the present study further collections of Barn Owl pellets were made for comparative purposes in 2003-06 and 2011-16 within three of the east Cork DEDs where, in 1992-94, the Bank Vole had not yet reached, or was only in the process of colonising (Smiddy & Sleeman 1994, Cooke *et al.* 1996). At two DEDs pellets were collected at single roost sites, while at the third DED pellets were collected at two nearby roost sites which because of their proximity have been treated here as one. For comparison, these methods are the same as applied in the 1992-94 study (Cooke *et al.* 1996).

To calculate the relative importance of each prey species in terms of biomass, mean weights were applied to each after Fairley and Smal (1988) (Brown Rat = 50.4 g; Field Mouse = 19.0 g; House Mouse = 16.5 g; Bank Vole = 18.1 g; Pygmy Shrew = 3.7 g; Common Frog *Rana temporaria* = 29.0 g). The three bats (species undetermined) were assigned a mean weight of 6.0 g after Smal (1987). Only one of the 40 birds could be specifically identified (Blackbird *Turdus merula* = 105.0 g), while the remaining 39 (all passerines) were classified as either 'small' (29 x 20.0 g) or large (10 x 70.0 g) based on the size of skeletal remains.

Although pellets were collected on an opportunistic rather than a systematic basis, most could be dated to a month of casting. Pellets were dried in ambient temperatures before analysis. During analysis each pellet was teased apart individually by hand and all bone and other hard material that might be useful for identification was retained. Mammal and Common Frog prey were identified using Yalden (2009). The total count of each mammal species being based on lower jaw bones, whichever side (left or right) gave the highest count. The habitat for approximately 2 km surrounding all roost sites (assessed by eye) was similar and consisted of intensive agricultural farmland with a mix of pastures, cereals and root crops. Hedgerow trees and patches of deciduous and coniferous woodland and scrub were also within the range of all roost sites, and a village with expanding housing was near one roost site (Cloyne).

Results

A total of 2,102 vertebrate prey items was recovered from Barn Owl pellets collected at three roost sites within three east County Cork DEDs in 2003-06 and 2011-16. Overall, the commonest species recorded (by number and biomass) were Field Mouse and Bank Vole. House Mouse and Pygmy Shrew were next in importance (by number) although in biomass the Pygmy Shrew was relatively insignificant. The largest species, the Brown Rat, was the third most important species in biomass. Bats, Common Frog and birds each formed low proportions of the diet (Table 1). Although there was some variation across the three DEDs, the same general hierarchical order pertained among the species (Table 1). Fewer than ten invertebrate prey items were recovered.

The Brown Rat appeared most frequently in the diet in October while the Field Mouse, House Mouse and Bank Vole appeared most frequently in late winter and spring. The Pygmy Shrew appeared in highest numbers in June and the Common Frog between February and June and most birds between January and June. The same general trends were evident at all three roost sites.

A comparison of data collected in 1992-94 (Cooke *et al.* 1996) with data collected in 2003-06 and 2011-16 for the present study shows the results of the colonisation of the area by the Bank Vole, which now accounts for about 29% of prey by numbers and biomass but ranging from about 18% to 42% in different DEDs (Table 1, Figure 1).

Discussion

As Barn Owl pellets for the present study were collected on an opportunistic rather than a systematic basis, assessment of seasonality of prey taken requires a cautious approach, although some results are in line with that noted in previous

Table 1. Percentage numbers and biomass of vertebrate prey taken by Barn Owls in 2003-06 and 2011-16 at the east County Cork District Electoral Divisions (DEDs) of Cloyne (CLOY), Dungourney (DGOU) and Ightermurragh (IGHM).

	% number	% biomass	% number	% biomass	% number	% biomass	% number	% biomass
DEDs	CLOY	CLOY	DGOU	DGOU	IGHM	IGHM	Overall	Overall
Brown Rat	7.2	19.3	3.6	10.5	9.1	24.1	6.7	18.3
Field Mouse	32.0	32.1	28.1	30.8	27.7	27.5	29.4	30.3
House Mouse	16.4	14.3	9.4	9.0	20.9	18.0	15.7	14.0
Bank Vole	29.4	28.2	40.1	41.9	18.6	17.6	29.3	28.7
Pygmy Shrew	13.2	2.6	16.5	3.5	17.6	3.4	15.6	3.1
Bat species	0.1	<0.1	0.0	0.0	0.3	0.1	0.1	<0.1
Common Frog	0.2	0.4	0.6	1.0	3.2	4.9	1.3	2.0
Bird	1.5	3.1	1.7	3.3	2.6	4.4	1.9	3.6
Total sample	**817**		**638**		**647**		**2,102**	

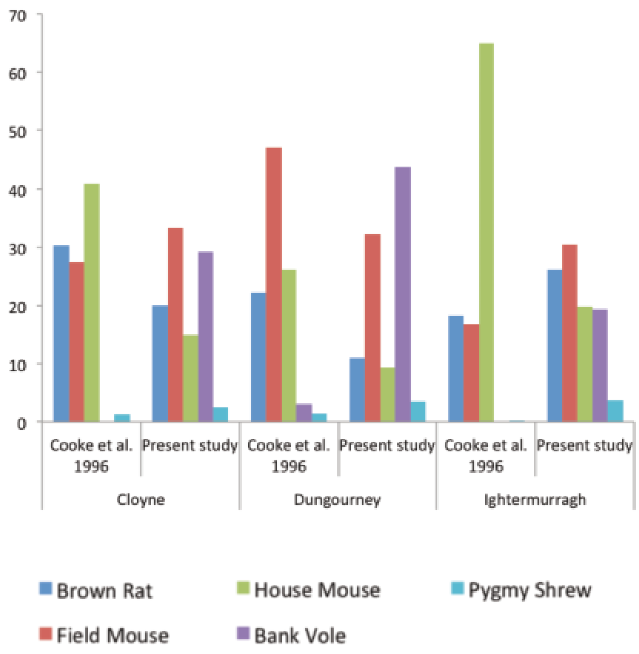

Figure 1. Percentage biomass of mammal prey taken by Barn Owls in 1992-94 (Cooke *et al.* 1996; n = 1,781) and in 2003-06 and 2011-16 (present study; n = 2,032) at the east County Cork District Electoral Divisions (DEDs) of Cloyne, Dungourney and Ightermurragh.

studies. The peak occurrence of the Brown Rat in autumn is presumed to result from the harvesting of crops suddenly exposing them (e.g. Fairley & Clark 1972) and of the Common Frog in spring is presumed to result from aggregations during spawning (e.g. Fairley & Clark 1972).

There was some variation in prey taken across the three DEDs, and between the results of the 1992-94 study (Cooke *et al.* 1996) and the present study (Table 1, Figure 1). The most obvious is the increase in the Bank Vole and the decline of the House Mouse at all DEDs; the House Mouse was found to be the most variable constituent of the prey in the 1992-94 study, occurring mainly around farmyards and strongly associated with 'other' crops, the largest component of which was fodder beet (Cooke *et al.* 1996). The Brown Rat declined at two DEDs, while the Field Mouse increased at two DEDs between the 1992-94 study and the present study (Figure 1). Apart from an increase in the Bank Vole, which colonised the area after 1992-94 (Smiddy & Sleeman 1994, Smiddy pers. obs.), there is no obvious explanation for some of the other changes in prey composition between 1992-94 and the present, or between individual DEDs.

Early studies on the diet of the Barn Owl in Ireland involved areas outside of the range of the Bank Vole (Smiddy *et al.* 2018). Most of these studies showed that the Field Mouse was the dominant species taken, with the Brown Rat second in importance in terms of biomass. Significant numbers of the Pygmy Shrew were taken in some studies (e.g. Fairley 1966, Fairley & Clark 1972, Clark 1974, Feehan 1995, Cooke *et al.* 1996) but, because of their small size, their contribution to prey biomass was low. In urban habitats the Brown Rat was dominant, and the House Mouse equalled or exceeded the Field Mouse in importance (Walsh 1984, 1985). Once the Bank Vole invaded an area the diet of the Barn Owl quickly switched to include significant numbers of that species (e.g. Smal 1987, Cooke *et al.* 1996, Farnsworth *et al.* 2002, O'Connell *et al.* 2006, Kelleher *et al.* 2010, Doyle *et al.* 2015). Following the recent arrival of the Greater White-toothed Shrew a further dietary switch appears to have taken place (Smiddy 2018), although this species has not yet invaded the present study area.

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Addressing the data dearth for Irish avifauna: biometrics of Skylarks *Alauda arvensis* inhabiting airfield grasslands

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In Ireland, large numbers of birds are ringed annually. However, although many bird-ringers diligently record biometric data, few published records describing the biometrics of Irish avifauna exist. Presently, ornithologists and other researchers regularly rely on biometric data obtained from other European populations when identifying the sex and age classes of birds. Yet, these data do not account for possible differential biometrics between disparate populations and sub-specific lineages. Moreover, sex class identification is often allocated based on a data threshold of polymodality, rather than through absolute confirmation of sex class. Here, biometrics of Skylarks *Alauda arvensis* inhabiting airfield grasslands in Counties Cork and Dublin are described. Molecular techniques are used to identify sex. We present a unique, if limited, dataset for Irish Skylarks, and encourage ringers and researchers to consider the value of their biometric data to the wider research community.

Introduction

Although large numbers of birds are ringed in Ireland every year (Tierney 2015), a paucity of studies describing the biometrics of the Irish avifauna exists (e.g. O'Halloran *et al.* 1992, Smiddy & O'Halloran 2015). However, while many bird-ringers regularly and meticulously collect basic biometric measurements, surprisingly few have examined these data. In most cases, biometric data have not been published for Ireland, and often remain inaccessible to most ornithologists and other researchers. Disappointingly, over the years, much data has undoubtedly been lost upon the retirement of ringers. Equally, other additional sources of biometric data,

Plate 12. Skylark (Dick Coombes).

such as cadavers or skins, appear to be rarely used (Kavanagh 1988).

Presently, published records may not appropriately reflect possible differential biometrics between disparate populations and sub-specific lineages. In some instances, demographic differences between available published estimates and new measurements obtained in the field may result in an incorrect allocation of sex or age class identification as data captured for continental birds may not necessarily encompass fringe populations and species clines. For example, northern and southern populations of Iberian Dippers *Cinclus cinclus* have shown morphological diversification in relation to the allometric relationship of body size and wing length. Birds inhabiting southern Iberia displayed longer wing lengths. Moreover, environmental factors were observed to predict mean tarsus length of Iberian Dippers, with tarsus increasing with increasing river slope and decreasing temperatures and precipitation (Arizaga *et al.* 2009).

The Eurasian Skylark *Alauda arvensis* is a widespread resident or short distance migrant species which breeds in most European countries (Cramp 1988, Donald 2004, Copland *et al.* 2012). Although often considered a common farmland bird, Skylark exhibits an on-going worldwide population decline, which has been predominantly linked to agricultural intensification (Donald *et al.* 2006, Praus & Weidinger 2015). Accordingly, the Skylark is designated as an Amber-listed Bird of Conservation Concern in Ireland (Crowe *et al.* 2010, Colhoun & Cummins 2013). Although still widespread, very few birds have been ringed in recent years in Ireland (annual mean of only seven during 2009–2013; Tierney 2015). This, in part, is likely reflective of lower numbers of active ringers and the difficulties of catching breeding Skylarks. Moreover, to a large extent, available biometric data for Skylarks relates to birds ringed in Scotland and England and European skins (Cramp 1988, Dougall 1997), often from relatively small sample sizes of < 100 specimens (Dougall 1997). Equally, in many instances the sex class for these samples was allocated based on polymodal distribution of biometric data, rather than with modern methods of identification, such as molecular sexing.

Collisions between birds and aircraft (bird strikes) are a consequence of modern aviation (Kelly & Allan 2006, Dolbeer *et al.* 2015). Although large-bodied birds can represent an air-safety hazard, efficient airfield habitat management can substantially reduce the risk of a bird strike event (Cleary & Dolbeer 2005). However, despite this, airfields often provide a habitat for some species of obligate grassland birds, such as the Skylark (Kelly & Allan 2006). The Skylark is not considered a threat to aviation safety, but birds can occasionally be struck by aircraft. Here, we report on biometrics of Skylarks struck by aircraft at two of Ireland's largest civil airports: Cork and Dublin International Airports.

Methods

Skylark carcasses were recovered from aircraft manoeuvring areas following reported strikes or during routine inspection at both Cork International Airport and Dublin International Airport. These mandatory inspections are designed to prevent Foreign Object Damage (FOD) to aircraft. Through the systematic examination of all aprons, taxiways and runways, these checks results in the removal of any debris located, including that composed of 'wildlife' (Kelly *et al.* 2017). Specimens were immediately placed in cold storage at -20 °C.

Morphological indicators were used to identify species and age of the birds by consulting the reference criteria of Cramp (1988) and Svensson (1992). For molecular sexing, DNA was extracted from individuals by placing a small sample (2 mm^3) of tissue (blood/tissue/feather) in 10% w/v chelex solution and incubating at 99°C for 70 minutes. The resulting supernatant was used as a template for PCR using the primers and conditions described in Griffiths *et al.* (1998) to determine the presence of two sex chromosomes in females or one in males. PCR was performed in duplicate for all samples to ensure that results were consistent. It was possible to age all birds as either adult (more than one year old) or juvenile (less than one year old). Standard measurements of wing (maximum chord, to the nearest 1 mm) were taken using a stopped rule, while a steel callipers was used to measure the tarsus, hind claw, bill (tip to distal corner of nostril), and head and bill. All were measured to the nearest 0.1 mm. Weights were recorded to the nearest 0.1 g using a digital balance (PESOLA® PTS3000). A Pearson correlation analysis was used to assess the relationship between male wing lengths and recovery month.

Results

In total, 54 usable Skylark carcasses were recovered from Cork (n = 9) and Dublin (n = 45) Airports over a 16-year period, 1999–2014. Molecular sexing was successfully performed for 47 of the Skylark carcasses, of which 39 (83 %) were male and eight (17 %) were female. Of these 47 sexed Skylarks, 46 specimens were aged as adult birds. The biometric data obtained from these 46 specimens are presented in Table 1, with further visual depictions of these data presented in Figures 1 and 2. A record of the date of death was obtainable for 41 specimens (Figure 1 a, b). Although a trend for increased wing length of male Skylarks from spring to autumn was observed, this is not statistically significant (Figure 1 c). Given that most specimens were collected in spring, summer and autumn months, and that wing lengths in the case of males did not significantly differ across the collection period (Figure 1 c), these birds are likely a representative sample of those inhabiting airfield grasslands. Equally, female wing lengths in relation to time of death suggest the birds are resident breeders.

Table 1. Measurements (mm) of adult Skylarks *Alauda arvensis* obtained from specimens recovered from Cork International and Dublin International Airports, showing mean ± SE, range, sample size (n), and location. M = male, F = female, CA = Cork Airport, DA = Dublin Airport. Total specimen n = 46; of which M = 38, F = 8.

Body Part	Sex	Mean ± SE	Range	Sample n	Location (n)
Wing	M	112.7 ± 0.6	102-119	34	CA (6), DA (28)
	F	105.8 ± 1.7	103-112	5	CA (0), DA (5)
Tarsus	M	28.2 ± 0.3	24.8-30.8	28	CA (3), DA (25)
	F	27.7 ± 0.4	25.7-28.3	6	CA (2), DA (4)
Hind claw	M	17.8 ± 0.4	11.8-22.3	36	CA (6), DA (30)
	F	16.0 ± 0.7	14.1-18.9	7	CA (2), DA (5)
Bill length	M	16.6 ± 0.3	13.4-19.0	25	CA (1), DA (2)
	F	16.7 ± 0.2	16.5-17.0	3	CA (3), DA (22)
Head and bill	M	35.1 ± 0.3	32.4-38.3	25	CA (4), DA (21)
	F	34.4 ± 0.1	34.2-34.5	3	CA (1), DA (2)

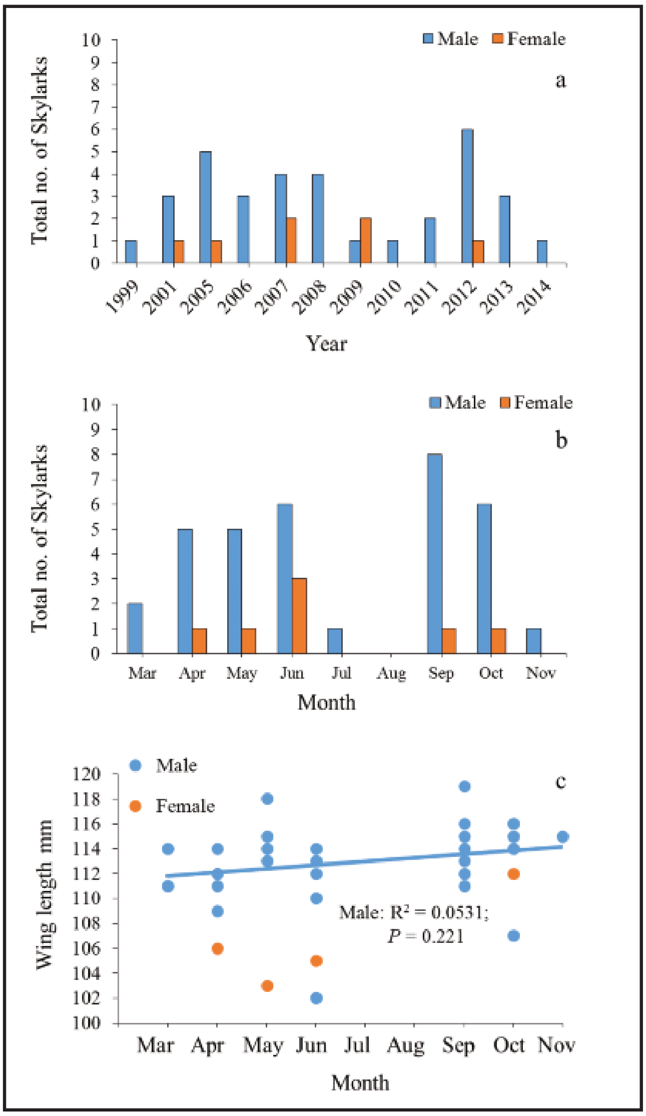

Figure 1. Total number of adult Skylark *Alauda arvensis* carcasses recovered from Cork International and Dublin International Airports, over the sixteen-year period 1999–2014. The number and sex of birds recovered each year (a: n = 41), per month (b: n = 41), and in relation to wing length per month (c: n = 40) is shown.

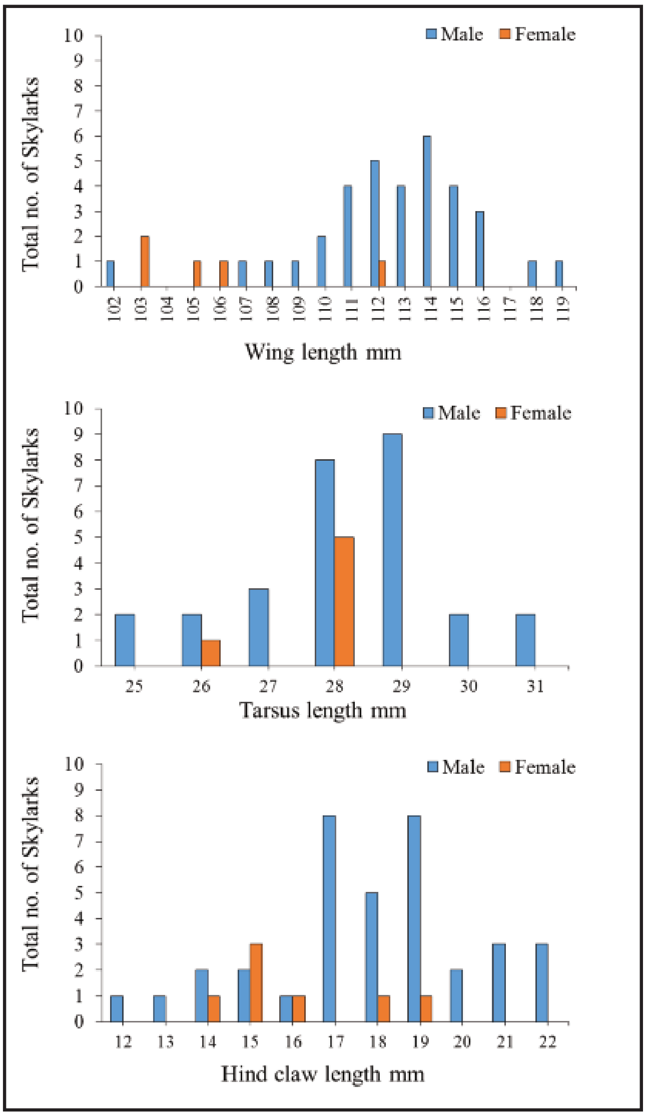

Figure 2. The measurement distributions, to the nearest millimetre, for wing (n = 39), tarsus (n = 34) and hind claw lengths (n = 43), taken from individual adult Skylark *Alauda arvensis* carcasses recovered from Cork International and Dublin International Airports, over the sixteen-year period 1999–2014.

Discussion

To our knowledge, the sex, age and biometrics of Skylarks killed at airfields have not been previously examined. Although our sample was predominantly male, Skylarks have previously shown sex- and age-biased mortality in collision with wind turbines. These turbine collisions mainly affected adult males (90.9%) and are therefore thought to relate to the male Skylarks characteristic breeding song-flights (Morinha *et al.* 2014). Due to the small sample size it has not been possible to make statistical comparisons between the sexes, or the examined populations. In addition, we do not present the weights, as many specimens were partially damaged. Although freezing and thawing of these specimens may have resulted in skin shrinkage, the wing length measurements obtained are within the ranges reported for live-caught Skylarks (see Dougall 1997).

The continued reluctance of ringers in Ireland to publish the results of their work has been noted on several occasions in recent decades (Hutchinson 1989, 1997, Smiddy & O'Halloran 2015). In many instances, this unpublished biometric data may be of great interest in the wider context of the species concerned at a European or even worldwide scale.

Plate 13. Skylark (Dick Coombes).

Biometric data, including wing length, are used to sex a wide variety of species based on measurement thresholds or mathematical estimates (Broughton & Clark 2017). However, enhanced understanding of age-dependant changes to biometrics, such as initial maturation and subsequent senescence, would enable improved sex and age determi-

Plate 14. Skylark (Michael O'Clery).

Plate 15. Skylark (Dick Coombes).

nation for many species (Piliczewski *et al.* 2018). In particular, the biometric aspects of senescence, whereby age-related decrease in performance can be mirrored in growth or quality of the renewable parts of the bird's body such as the feathers or bill sheath, have only occasionally been examined (Piliczewski *et al.* 2018).

The importance of incidental data collection, such as observation of apparent novel behaviours (e.g. Ryan *et al.* 2016) and occasional chance events (e.g. Kelly *et al.* 2016) can improve our understanding of species' life histories, ecological processes and anthropogenic impacts (Coughlan *et al.* 2017). Bird-ringers, birdwatchers and photographers are uniquely placed to contribute to the collection of these additional data. Accordingly, wildlife enthusiasts should be encouraged to work with research groups to better catalogue meaningful data. Citizen science initiatives may help increase the collection and cataloguing of such observations (Coughlan *et al.* 2017). Overall, as argued by Smiddy and O'Halloran (2015), the importance of short term studies, and of limited and opportunistically collected data are emphasised. Even if there is no immediate prospect of expanding on the work, future ornithologists and the study species may benefit.

Finally, the sub-specific status of the 'Irish' Skylark requires further investigation. For example, Donald (2004) refers to the subspecies in Ireland as *Alauda arvensis scotica* (♂ wing length <114 mm; ♀ $\leq$ 104), which can also be found in the Faroes, western Scotland and northwest England. However, the nominate form *A.a. arvensis* (♂ wing length >114 mm; ♀ $\geq$ 105 mm) is known from northern Europe, including England and southern Scotland. In addition, Ruttledge (1975) refers to Skylarks "collected in the west of Ireland" as *A.a. theresae* the darkest form of the species. Ruttledge (1975) also refers to specimens of an eastern race *A.a. intermedia* ("of southern, central and eastern Europe") as having been taken three times at Irish light stations. However, neither Donald (2004) nor Parkin and Knox (2010) mention *intermedia*. Additionally, Donald (2004), who examined many specimens including those available in museums throughout Europe, emphasises the very high level of morphological variation, even within local populations. Accordingly, a genetic study of Skylarks might prove to be highly informative.

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Barn Owl *Tyto alba* diet in Ireland: a review

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The number of small mammal species available as prey to Barn Owls *Tyto alba* in Ireland is limited nationally to four, with two others present in restricted areas. Twenty nine studies (1964-2018) on diet were reviewed and it was found that numbers of Field Mice *Apodemus sylvaticus* and Pygmy Shrews *Sorex minutus* taken declined significantly while numbers of Bank Voles *Myodes glareolus* taken increased significantly. There was no significant change in numbers of Brown Rats *Rattus norvegicus* and House Mice *Mus domesticus* taken, although there was a downwards trend for both. The Greater White-toothed Shrew *Crocidura russula* has occurred in Barn Owl diet in recent years, and it will probably increase in importance with future range expansion. Bats (Chiroptera), Common Frog *Rana temporaria* and birds (Aves) are taken either rarely or in relatively small numbers.

Plate 16. Barn Owl pellets (John Lusby).

Introduction

The Barn Owl *Tyto alba* was once widespread in Ireland, but a range decline of 47% has occurred since the 1960s (Balmer *et al.* 2013). It is now Red-listed on Birds of Conservation Concern (Colhoun & Cummins 2013). Several probable reasons for its decline have been proposed, most associated with landscape changes due to agricultural intensification (e.g. removal of hedgerows, decline in small-scale tillage, switch from hay- to silage-making, increased use of pesticides, herbicides and rodenticides, loss of nest sites through removal of hedgerow trees) and renovation or demolition of old buildings. Increased mortality on some major road systems further adds to its poor conservation status (Lusby & O'Clery 2014). It is likely that the decline has not been caused by any one factor, but perhaps by several acting in combination; it has been argued that mortality due to rodenticides was insufficient on its own as an explanation (Newton 2017). The Irish agricultural landscape has changed immeasurably over the last two and a half centuries from one based on manual and horse-drawn practices (Bell & Watson 2008) to one of mechanised intensive monocultures of cereals, grass, forage crops and vegetables, especially since accession to the European Union in 1973 (Buttimer *et al.* 2000, Feehan 2003, Aalen *et al.* 2011). It is perhaps unsurprising that the Barn Owl population would decline in the face of such large-scale changes within its main habitats (Cooke *et al.* 1996, Shawyer 1998, Love *et al.* 2000).

The main prey of the Barn Owl throughout its range consists of small mammals, especially *Microtus* voles wherever they occur (Bunn *et al.* 1982, Taylor 1994, Toms 2014). There are few small mammal species in Ireland, but those that are present are especially important for driving ecosystem function. Diurnal hunting by Barn Owls in Ireland is rare and likely reflective of the predominance of the nocturnal Field Mouse *Apodemus sylvaticus* as prey, compared to Britain, where it is taken less frequently (Love *et al.* 2000). There is variation in diurnal hunting between individuals in Britain, with weather, prey availability and remoteness from human populations also possible factors (Toms 2014). Small mammals are major prey items for predatory birds and mammals and they are themselves important predators on plant seeds, insects, spiders and mites. Small mammals also harbour and transmit diseases, damage property, crops and stored products, and they often cause distress to humans when they occur near or in buildings.

The small mammal populations which support Barn Owls are not static and have changed significantly in recent decades (Fairley 1984, 2001). Ireland has only four indigenous mammals small enough to be killed by Barn Owls (Brown Rat *Rattus norvegicus,* Field Mouse, House Mouse *Mus domesticus* and Pygmy Shrew *Sorex minutus*). This prey base has been increased by two recently introduced species (Bank Vole *Myodes glareolus* and Greater White-toothed Shrew *Crocidura russula*) (Claassens & O'Gorman 1965, Tosh *et al.* 2008). While these new species may, in the short-term, provide prey for Barn Owls and other predators (e.g. Doyle *et al.* 2015, Smiddy 2017), there is uncertainty about their long-term effects on indigenous small mammal species already present. For example, there is evidence that the presence of the Greater White-toothed Shrew has a positive effect on the abundance of Bank Voles, but a negative effect on the abundance of Field Mice and the occurrence of Pygmy Shrews (Montgomery *et al.* 2012). The phenomenon where the presence of one recently introduced species facilitates another and compounds negative effects on indigenous species has been termed 'invasional meltdown' (Montgomery *et al.* 2012, 2015).

In the absence of data on the distribution and abundance of small mammal prey species, published studies of Barn Owl diet can be examined to provide information on what species they were feeding on in the past. This can give a measure of population trends, assuming the owls take prey that is most abundant. There is evidence of both positive and negative correlations between small mammal prey and owl diet selection; Hanney (1962) observed an inverse relationship between prey and diet, while a positive relationship between availability of bats and Tawny Owl *Strix aluco* diet was observed in Poland (Lesiński *et al.* 2008). There is evidence from the present review that Barn Owls prey intensively on the invasive Bank Vole (e.g. Doyle *et al.* 2015) and Greater White-toothed Shrew (e.g. Smiddy 2018a) once these species become available.

Do owls select these species in preference to indigenous ones, have the indigenous species declined because of the presence of the invasive ones, or are there other reasons? These questions have important implications for predators, pest control, seedling survival and public health. Brown Rats are also significant reservoirs and vectors of disease (Fairley 2001), but so too are Bank Voles, for example, hantavirus in Sweden (Olsson *et al.* 2005) and Greater White-toothed Shrews and *Leptospira* in Ireland (Nally 2016a, 2016b).

Analysis of owl pellets provides a method of monitoring small mammals (Strachan 1995, Yalden 2009) and occasionally the discovery of a new species, such as the Greater White-tooted Shrew (Tosh *et al.* 2008). Some incidental observations on the diet of the Barn Owl in Ireland have been made (e.g. Thompson 1849, Ussher & Warren 1900, Kennedy *et al.* 1954), but quantitative data are available only since the mid-1960s (Fairley 1966). In this paper we examine long-term (1964-2018) changes in prey selection in the Barn Owl, especially with reference to studies in the period following the introduction of the Bank Vole and Greater White-toothed Shrew.

Methods

Twenty-nine studies have been included in this review (listed in Appendix 1 and indicated by an asterisk in the reference section). Twenty-three of these contain data relating to over 100 vertebrate prey items with four reporting data for over 5,000, and the largest (Cooke *et al.* 1996) for almost 11,000. Invertebrate prey has not been included in this review as this is a difficult-to-quantify element and is generally considered an insignificant part of Barn Owl diet (Bunn *et al.* 1982, Shawyer 1998, Love *et al.* 2000, Toms 2014); invertebrate prey (mostly beetles, Coleoptera) has been reported in 28% of the sources analysed. However, it has been shown that the consumption of invertebrates by Barn Owls across Europe has strongly declined since 1860, no doubt related to a reduction in their diversity (Roulin 2016a).

Some publications did not give totals for individual prey items (e.g. Fairley & Smal 1989), but totals were calculated based on the overall numbers and percentages given. It was not possible to distinguish Irish data from the published account in Glue (1974), but the British Trust for Ornithology (BTO) supplied the raw data for Ireland. Glue (1974) did not mention Irish studies in his reference section, but the raw data makes it clear that he used information from Counties Down, Fermanagh and Galway which had already been published (Fairley 1966, Fairley & Deane 1967, Fairley & Clark 1971), and information from Counties Antrim and Donegal which had not; the latter have been incorporated into the dataset for the present paper.

Inconsistencies in the number of prey items given in some sources (e.g. Feehan 1995) have been corrected. In one paper (Smal 1987) no distinction was made between Field Mice and House Mice in some pellet collections from one site.

To resolve this a proportion for each species was calculated based on the known proportions of the two species in other pellet collections from the same site. Another source (Eadsforth *et al.* 1996) provided combined prey data from five counties, two of which had Bank Voles and three of which did not, therefore its usefulness was limited in any examination of prey taken outside and within the range of this vole (a paper by Harrison *et al.* (1990) is a preliminary report on that of Eadsforth *et al.* (1996), therefore only the latter is referred to hereafter). There was an indication that some data may have been used in more than one paper (Sleeman & Kelleher 2008 and Kelleher *et al.* 2010; Ronayne *et al.* 2011 and Ronayne & Sleeman 2013), so in these cases we used the minimum number of prey items recorded (see Appendix 1).

Prey biomass for rodents, Pygmy Shrew and Common Frog *Rana temporaria* was calculated from Fairley and Smal (1988), for Greater White-toothed Shrew (11 g) from McDevitt (2016) and for bats (6 g) from Smal (1987). Most birds were not specifically identified, and mean weights were applied referring to data or indications of size reported in the source publications (Appendix 1). The biomass of birds identified to species level was taken from *Birds of the Western Palearctic* (1983-1994, Volume 3-8). Some early studies of Barn Owl diet in Ireland applied correction factors when comparing the proportional contribution to the diet made by each prey species to account for differences in sizes between them (Southern 1954). This method was refined by later authors who realised the correction factors had shortcomings and had been developed for Tawny Owl studies (which does not occur in Ireland) and therefore did not relate to Irish material (e.g. Smal 1987). It is likely that early studies underestimated the contribution to diet made by birds when there were many (>10) in the sample since the correction factor applied was 20 g per bird where the species was not identified (e.g. Fairley & Clark 1972, Clark 1974, Glue 1974). It has been shown in subsequent studies that mean weights of birds are usually much higher (e.g. Walsh 1984 (53 g), Smal 1987 (51 g), Fairley & Smal 1989 (59-69 g), O'Connell *et al.* 2006 (64-71 g)). Weights of individual birds can be calculated from humerus length (Morris & Burgis 1988). However, while it has been possible to apply new correction factors to mammal prey, no retrospective recalculation of bird weights has been carried out since some studies either gave no indication of size or referred to size in terms of 'small', 'medium' or 'large' (see Appendix 2).

All prey data were assigned to a 5-year period when the pellets were produced based on the information provided within each publication. However, a minority of studies spanned more than one 5-year period, and where the author(s) did not provide information on the number of prey items recorded year by year, prey data were assigned to the 5-year period to which the greater part of the study related. The data relating to Barn Owl diet in Ireland comes from 19 counties over a period of about 50 years. The quantity of data available varies from 14 prey items (Donegal) to 21,959 prey items (Cork) (see Appendix 3). The disparity in the quantity of data available temporally, geographically, seasonally and between habitats and areas outside and within the ranges of the Bank Vole and Greater White-toothed Shrew has placed some constraints on the extent of the analysis undertaken. We present frequency distributions of vertebrate prey (numbers and biomass) outside and within the ranges of the Bank Vole and Greater White-toothed Shrew (Tables 1, 2, 3), and for rodents and shrews in 5-year periods between 1964-68 and 2014-18 (Figures 1, 2). Temporal trends in proportions (after arcsine-transformation of percentage values) of the four rodents and Pygmy Shrew for each 5-year period were examined using linear models for each prey species where the response variable was the proportion of each species in the diet over the eleven 5-year study periods, with study period included as a continuous variable (Table 4). Statistical analyses were performed using R version 3.4.3 (www.r-project.org).

Results

A total of 46,605 vertebrate prey items was recorded in the 29 dietary studies reviewed. The Field Mouse (20,841) was the commonest species, followed by the Bank Vole (8,688), Pygmy Shrew (6,845), House Mouse (4,126), Brown Rat (3,989) and birds (1,195). All other species were recorded in much smaller numbers (Appendix 1). In areas where the Bank Vole was absent the order of importance (numbers) was Field Mouse, Pygmy Shrew, House Mouse and Brown Rat (Table 1), while in areas where the Bank Vole was present the order of importance was Field Mouse, Bank Vole, Pygmy Shrew and Brown Rat (Table 2). Many more Field Mice, House Mice and Pygmy Shrews were taken in the absence of the Bank Vole (Tables 1, 2). When these data were converted to biomass the

Table 1. Composition of Barn Owl prey species in areas outside the range of Bank Vole (n = 14,565). Data refer to percentage numbers and biomass.

Species	Number	% number	% biomass
Brown Rat	1,103	7.6	21.0
Field Mouse	7,439	51.1	53.3
House Mouse	1,887	12.9	11.8
Bank Vole	0	0.0	0.0
Pygmy Shrew	3,467	23.8	4.8
Greater White-toothed Shrew	0	0.0	0.0
Bats	33	0.2	0.1
Common Frog	130	0.9	1.4
Birds	506	3.5	7.6

Table 2. Composition of Barn Owl prey species in areas within the range of Bank Vole (n = 30,545). Data refer to percentage numbers and biomass.

Species	Number	% number	% biomass
Brown Rat	2,749	9.0	22.3
Field Mouse	13,075	42.8	40.0
House Mouse	2,064	6.8	5.5
Bank Vole	8,391	27.5	24.5
Pygmy Shrew	3,282	10.7	2.0
Greater White-toothed Shrew	0	0.0	0.0
Bats	64	0.2	0.1
Common Frog	279	0.9	1.3
Birds	641	2.1	4.3

Brown Rat gained in importance (because of its large size), while the Pygmy Shrew declined (because of its small size), but the Field Mouse held its high position (Tables 1, 2). When both invasive species were present together (although based on a small sample) the Greater White-toothed Shrew (54%) and Bank Vole (22%) became the most important species (biomass), while all others declined in importance (Table 3). The increase in Bank Voles in the diet of the Barn Owl since the 1960s has been significant, and the decline in Field Mice has been equally so. There has also been a significant decline in Pygmy Shrews (Figure 1, Table 4). No significant change was detected in Brown Rats and House Mice, although the trend was downwards for both (Figure 1, Table 4). A similar trend was observed when these data were converted to biomass (Figure 2). The 97 bats recorded constituted 0.21% of the overall total of vertebrate prey (Appendix 1). Common Frogs were recorded also in low numbers (0.91%) while birds formed 2.56% of total prey numbers (Appendix 1).

Table 3. Composition of Barn Owl prey species in areas within the combined ranges of Bank Vole and Greater White-toothed Shrew (n = 576). Data refer to percentage numbers and biomass.

Species	Number	% number	% biomass
Brown Rat	14	2.4	8.7
Field Mouse	57	9.9	13.4
House Mouse	6	1.0	1.2
Bank Vole	98	17.0	22.0
Pygmy Shrew	1	0.2	<0.1
Greater White-toothed Shrew	399	69.3	54.4
Bats	0	0.0	0.0
Common Frog	0	0.0	0.0
Birds	1	0.2	0.2

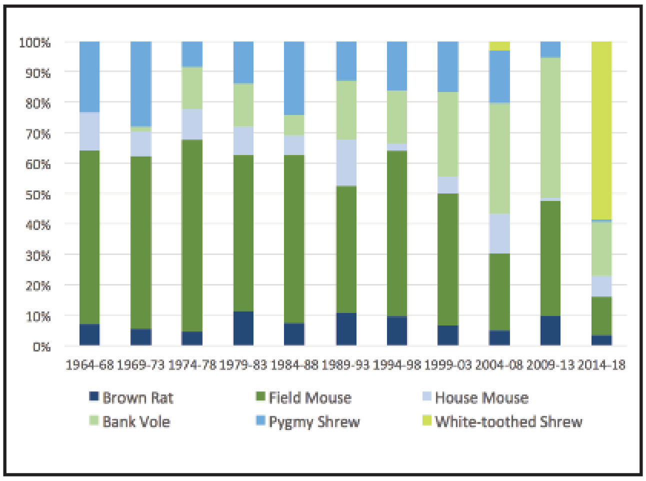

Figure 1. Proportional composition of rodents and shrews taken by Barn Owls in Ireland in 5-year periods (n = 849; 2,682; 1,722; 7,064; 8,365; 12,756; 584; 2,131; 1,738; 6,407; 590 individual prey items in each 5-year period, respectively). Data are expressed as percentage numbers.

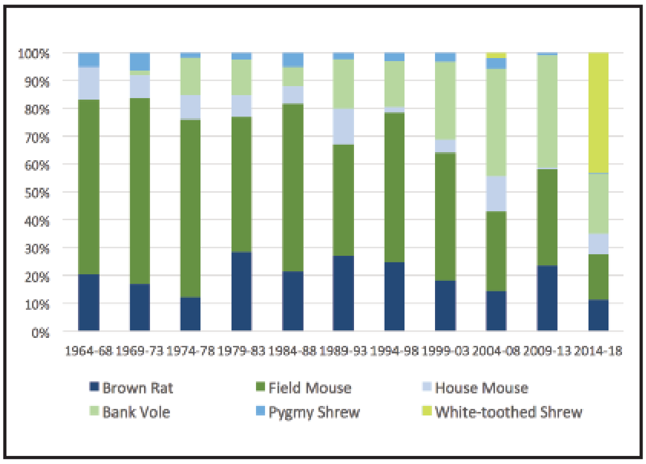

Figure 2. Proportional composition of rodents and shrews taken by Barn Owls in Ireland in 5-year periods (n = 849; 2,682; 1,722; 7,064; 8,365; 12,756; 584; 2,131; 1,738; 6,407; 590 individual prey items in each 5-year period, respectively). Data are expressed as percentage biomass.

Table 4. Linear models evaluating temporal trends in the proportion of each prey species in Barn Owl diet across eleven 5-year study periods.

Species	Estimate	SE	T value	P value
Brown Rat	-0.001	0.003	-0.262	0.799
Field Mouse	-0.039	0.008	-4.699	0.001**
House Mouse	-0.005	0.004	-1.372	0.203
Bank Vole	0.034	0.008	3.980	0.003**
Pygmy Shrew	-0.016	0.006	-2.598	0.029*

Notes: * = significant change; ** = highly significant change.

Discussion

To our knowledge, this is the most comprehensive review of Barn Owl diet in Ireland to date, although short generalised reviews have appeared previously (Fairley 1984, 2001). Owls in general, and Barn Owls in particular, have been described as highly efficient predators of rats and mice in Ireland (Fairley 2001), and popular literature refers to them as 'the farmer's friend' because of this. Irish Barn Owls have a limited range of prey species on which to feed compared to Barn Owls elsewhere (Bunn *et al.* 1982), and this makes their diet of significant ecological interest. Important and relevant too is the fact that the Barn Owl is a declining species (Balmer *et al.* 2013), and that two new prey species (Bank Vole and Greater White-toothed Shrew) have colonised Ireland in recent decades (Claassens & O'Gorman 1965, Tosh *et al.* 2008).

Although there are few small mammals in the Irish environment, surprisingly little is known about population sizes, fluctuations and general dynamics of different species in different habitats, and at different times. This problem is not unique to Ireland and the absence of survey data on supposedly 'common' species has been commented upon elsewhere (e.g. Matthews *et al.* 2018); obtaining such data involves laborious work, often over a long period (Zárybnická *et al.* 2017). Long-term analysis of Barn Owl pellets is one potential means of obtaining such data (e.g. Yom-Tov & Wool 1997, Torre *et al.* 2015, Szép *et al.* 2017), but whether it is possible or not is still debated (Meek *et al.* 2012). The degree to which Barn Owls take their prey selectively or in an opportunistic way is also uncertain (Taylor 1994, Tores *et al.* 2005, Bernard *et al.* 2010). Clearly, this is relevant to the validity of using dietary studies as a means of monitoring small mammal populations.

Plate 17. Male Barn Owl at a nest site in Co. Kerry (Michael O'Clery).

This review shows that Irish Barn Owls take all terrestrial small mammal species available to them (Appendix 1), but the degree to which they are selective is unknown (Hanney 1962). The emphasis in many Irish studies has been the question of which species are taken most frequently outside and within the restricted range of the Bank Vole (Tables 1, 2). A decline in importance for the Field Mouse and Pygmy Shrew, exacerbated further within the combined ranges of the Bank Vole and Greater White-toothed Shrew (Table 3), is in accordance with species-replacement predictions relating to the effects of invasive species on indigenous ones (Montgomery *et al.* 2012, McDevitt *et al.* 2014). In Europe, the number of insectivorous small mammals (shrews and moles) in the diet of Barn Owls has declined over the last 150 years (Roulin 2016b).

The invasive Bank Vole and Greater White-toothed Shrew appear to be preyed upon by Barn Owls as soon as they are available, and they may quickly become a significant prey item, or even the most important one numerically (e.g. Smal 1987, Cooke *et al.* 1996, Farnsworth *et al.* 2002, O'Connell *et al.* 2006, Foley & Sleeman 2008, Kelleher *et al.* 2010, Doyle *et al.* 2015, Smiddy 2018a, 2018b). The quantity of data on the Greater White-toothed Shrew available for this review is limited, and significant datasets remain unpublished (Lusby & O'Clery 2014).

The Brown Rat and House Mouse both live as commensals of humans, and they can be significantly affected by rodenticide use (e.g. Eadsforth *et al.* 1996). It has been suggested that the Brown Rat population may have decreased in recent years (e.g. O'Shea *et al.* 2010), but the data presented here on Barn Owl diet does not support this suggestion, albeit numbers in pellets showed a non-significant decline through time (Figure 1, Table 4). Because of its large size, the Brown Rat forms around 21% of biomass outside and within the range of the Bank Vole, but only 9% within the combined ranges of the Bank Vole and Greater White-toothed Shrew (Table 1, 2, 3), although caution is required as the latter sample is small. It should perhaps be considered that the importance of the Brown Rat may have been somewhat inflated in the past by studies at urban areas in Waterford (Walsh 1984) and Mitchelstown (Smal 1987) where the species was common. There have been no major studies of Barn Owl diet in such habitats in the last 20 years.

Some authors have stated that Brown Rats are the main prey of Irish Barn Owls (e.g. Macdonald & Barrett 1993), which is contrary to our extensive review (Figures 1, 2). There is a popular misconception that owls 'control' rat numbers, but this is hardly possible since a bird weighing 280-450 g is unlikely to dispatch a full-grown rat of breeding age at 300-500 g. Indeed, most Brown Rats taken by Barn Owls weigh little more than 50 g (Morris 1979, Fairley & Smal 1988) and are too young to breed, therefore rather than 'controlling', the owls are taking the 'interest' rather than the 'capital' among the rat population, despite their significance in terms of biomass (Figure 2). However, data from Rome (Salvati *et al.* 2002) suggests rats are an important food item during the owls' breeding season. This may be the case also in Ireland, perhaps in parts not yet colonised by Bank Voles, or in urban areas.

The House Mouse has been the most variable prey species (numerically) in the diet of the Barn Owl with some studies reporting few and others many (e.g. Cooke *et al.*

Plate 18. A collection of jawbones from one Barn Owl pellet in Co. Tipperary in 2016, containing remains of several Greater White-toothed Shrews and a single Field Mouse (Michael O'Clery).

1996). A study in County Cork in the 1990s showed numbers at different sites varying from 1-75%, they occurred mainly around farmyards and their numbers were strongly correlated to land use (fodder crops) (Cooke *et al.* 1996). The numbers taken by Barn Owls appear to be declining, but the trend is not statistically significant (Figure 1, Table 4) (but see above for Brown Rat).

Bats are rare prey in Irish Barn Owl dietary studies (0.21%) (Appendices 1, 2). In Europe, they are also rarely taken (0.12%), and this occurs more frequently on islands and in the east and south. It might be expected that greater numbers of bats would be taken since both may occur near each other in ruined buildings, where some owls are known to specialise on them, apparently taking mostly young in an opportunistic way (Petrželková *et al.* 2004, Roulin & Christe 2013). Predation on bats has decreased across Europe during the last 150 years, which may reflect historical declines in bat populations (Roulin & Christe 2013). Common Frogs are also rarely reported as prey in Ireland (0.91%) (Appendix 1) as is the case also in Europe as a whole (0.54%) (Roulin & Dubey 2013). Most Irish studies reporting frogs indicate a peak of consumption in spring. Frogs gather at ponds to spawn where they apparently become more susceptible to predation, and this also appears to be true elsewhere (Roulin & Dubey 2013). Birds are taken regularly in Ireland (2.56%), usually in small numbers, although in a few cases the number taken was quite high (e.g. 31.34% in Wexford) (Fairley & Smal 1989). The birds taken most frequently are often flocking species (e.g. Common Starling *Sturnus vulgaris* and House Sparrow *Passer domesticus*) which perhaps roost near to owl roost sites, and winter migrants such as Redwing *Turdus iliacus* (Appendices 1, 2). The percentage of birds in the diet of Barn Owls across Europe (2.43%) (Roulin 2015) is close to that reported here for Ireland.

As we have seen, Barn Owl prey across Europe appears to reflect long-term declines of insects and their predators such as bats, birds and terrestrial mammal insectivores (shrews) (Roulin & Christe 2013, Roulin 2015, 2016a, 2016b). The study of Barn Owl prey in Ireland has resulted in the detection of invasive mammals and in expansion of their ranges, and this may be useful in reflecting changing small mammal population densities of the past, present and the future.

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Appendix 1

Total numbers of vertebrate prey taken by Barn Owls in Ireland based on analysis of 29 studies. It should be noted that these percentages do not imply relative abundance of each species in the environment, rather they reflect the number of studies carried out in different habitats outside and within the range of the Bank Vole and Greater White-toothed Shrew. Sources of data are shown in notes below.

Species	Number	Percentage
Brown Rat *Rattus norvegicus*	3,989	8.56
Field Mouse *Apodemus sylvaticus*	20,841	44.72
House Mouse *Mus domesticus*	4,126	8.85
Bank Vole *Myodes glareolus*	8,688	18.64
Pygmy Shrew *Sorex minutus*	6,845	14.69
Greater White-toothed Shrew *Crocidura russula*	399	0.86
Rabbit *Oryctolagus cuniculus*	1	<0.01
Bats (Chiroptera)	97	0.21
Common Frog *Rana temporaria*	422	0.91
Birds (Aves)	1,195	2.56
Fish (Pisces)	2	<0.01
Totals	**46,605**	**100.00**

Notes: All published and accessible unpublished literature on Barn Owl diet, grouped into three categories; (1) studies outside of the range of the Bank Vole (Fairley 1966, Fairley & Deane 1967, Fairley & O'Gorman 1971, Fairley & Clark 1971, 1972, Clark 1974, Glue 1974, Walsh 1984, 1985, Smal 1987, Fairley & Smal 1989, Feehan 1995, Cooke *et al.* 1996, Eadsforth *et al.* 1996, Foley *et al.* 2006, (2) studies within the range of the Bank Vole (Fairley & Forster 1974, Forster & Fairley 1975, Walpole 1977, Walsh 1985, Smal 1987, Fairley & Smal 1989, Cooke *et al.* 1996, Eadsforth *et al.* 1996, Farnsworth *et al.* 2002, O'Connell *et al.* 2006, Foley & Sleeman 2008, Sleeman & Kelleher 2008, Kelleher *et al.* 2010, Ronayne *et al.* 2011, Ronayne & Sleeman 2013, Doyle *et al.* 2015, Smiddy 2018b) and (3) studies within the combined ranges of the Bank Vole and Greater White-toothed Shrew (Tosh *et al.* 2008, Smiddy 2018a). One reference (Patterson 1908) purporting to relate to the Barn Owl was shown to involve the Long-eared Owl *Asio otus* (Fairley 1992) and is therefore excluded from this review.

Appendix 2

Bird and bat species taken by Barn Owls in Ireland based on analysis of 29 studies; for Corncrake see Kennedy *et al.* (1954).

Species
Corncrake *Crex crex*
Dunlin *Calidris alpina*
Common Snipe *Gallinago gallinago*
Woodcock *Scolopax rusticola*
Goldcrest *Regulus regulus*
Blue Tit *Cyanistes caeruleus*
Great Tit *Parus major*
Skylark *Alauda arvensis*
Barn Swallow *Hirundo rustica*
Wren *Troglodytes troglodytes*
Common Starling *Sturnus vulgaris*
Blackbird *Turdus merula*
Fieldfare *Turdus pilaris*
Song Thrush *Turdus philomelos*
Redwing *Turdus iliacus*
European Robin *Erithacus rubecula*
Dunnock *Prunella modularis*
House Sparrow *Passer domesticus*
Tree Sparrow *Passer montanus*
Meadow Pipit *Anthus pratensis*
Chaffinch *Fringilla coelebs*
Greenfinch *Carduelis chloris*

Lesser Horseshoe Bat *Rhinolophus hipposideros*
Daubenton's Bat *Myotis daubentonii*
Natterer's Bat *Myotis nattereri*
Leisler's Bat *Nyctalus leisleri*
Pipistrelle Bat *Pipistrellus* species
Long-eared Bat *Plecotus auritus*

Appendix 3

Counties in which studies of Barn Owl diet have taken place and the numbers of vertebrate prey taken in each (note that some studies involved several counties). One study (Eadsforth *et al.* 1996) encompassed five counties (*), but the authors did not indicate how many of the 917 prey items was taken in each (n = 29 studies).

County	Number of prey	Number of studies
Antrim	430	1
Carlow	306	1
Clare	22	1
Cork*	21,959	12
Donegal	14	1
Down	650	1
Dublin	140	1
Fermanagh	205	1
Galway	7,764	7
Kerry*	8,437	3
Kildare	396	1
Kilkenny*	?	1
Limerick	253	3
Offaly	3,077	1
Tipperary	67	1
Waterford*	1,178	4
Westmeath	419	1
Wexford*	67	2
Wicklow	304	1

An account of Purple Martin *Progne subis* records from Ireland and Britain during the 19th century

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During 1839-40, a specimen of Purple Martin *Progne subis* was reported to have been shot near Kingstown (Dun Laoghaire), Co Dublin. Although the identification of the Kingstown specimen, which is preserved in the collections of the National Museum of Ireland – Natural History (NMINH), has never been questioned, the authenticity of this single Irish record has long been doubted. Six more specimens of Purple Martin were reported from Britain between 1842 and 1878, but their authenticity has also been doubted. However, the recently confirmed observation of a juvenile Purple Martin on the Isle of Lewis (Scotland) during 2004 and a further two juveniles on the Flores and Corvo (Azores) during 2004 and 2011 respectively, proves beyond doubt that this Nearctic hirundine can occur naturally in Europe. The current review collated and assessed the historical information available on the Irish and British 19th century records of Purple Martin, and in the light of recently confirmed sightings in the Western Palearctic, it is proposed that a revaluation of some of these records is warranted.

Introduction

According to Parkin and Knox (2010), the nominate race of the Purple Martin *Progne subis* breeds more or less continuously across eastern North America, from the Great Lakes to the Gulf of Mexico, with an extension into Alberta. There is a series of discontinuous populations elsewhere, ranging from the southern tip of British Columbia to northern Baja California *(arboricola)* and thence through central Mexico *(hesperia)*. Strongly migratory, the Purple Martin winters in the lowlands of South America, reportedly most migrating through Central America, although recorded commonly in Bermuda. Turner and Rose (1989) noted that Purple Martins have been recorded casually north to the Pribilof Islands (Alaska), central Yukon, northwestern Ontario and northern Nova Scotia, and that individuals have appeared in the British Isles.

The first known record of Purple Martin from Europe was reported to have been shot near Kingstown (Dun Laoghaire), Co. Dublin (Yarrell 1840), on the east coast of Ireland, most likely during the autumn of 1839 (Parkin & Knox 2010). However, the authenticity of this single Irish specimen of Purple Martin has long been doubted. It is still included in the Irish List of Birds (IRBC, http://www.irbc.ie/) under Category D1, indicating species that would otherwise appear in Categories A or B except that there is reasonable doubt that they have ever occurred in a natural state.

Plate 19. Specimen of female Purple Martin *Progne subis* shot near Kingstown (Dun Laoghaire), Co Dublin during 1839-40 [NMINH:2003.52.1011 (DE-746)] (Paolo Viscardi).

Plate 20. Specimen of female Purple Martin *Progne subis* shot near Kingstown (Dun Laoghaire), Co Dublin during 1839-40 [NMINH:2003.52.1011 (DE-746)] (Paolo Viscardi).

Although six specimens of Purple Martin were subsequently reported from Britain between 1842 and 1878, their authenticity has also been doubted. However, the confirmed observation of a juvenile on the Isle of Lewis, Outer Hebrides (Scotland) on 5 September 2004 (Coyle *et al.* 2004, 2007) led to the admission of the species under Category A in the most recent Checklist of the Birds of Britain (McInerney *et al.* 2018). A second juvenile was also confirmed from the Flores (Azores) on 6 September 2004 (Coyle *et al.* 2007), the day after the Isle of Lewis individual was initially observed. The sighting of a third juvenile was confirmed from Corvo (Azores) between 13 and 18 October 2011 (Matias *et al.* 2012, Monticelli *et al.* 2015, Alfrey *et al.* 2018).

In the editorial remarks appended to the account of the Isle of Lewis record (Coyle *et al.* 2007), Bob McGowan (Chairman of the British Ornithologists' Union Records Committee) commented: 'The two confirmed Western Palearctic records in 2004 lends veracity to the earlier report of two juvenile birds on Pico (Azores) on 28 September 1996 (Coyle *et al.* 2007) and perhaps also to the Dublin (Kingstown) record, viewed sceptically at the time because transatlantic vagrancy by Purple Martin was considered improbable'.

The current review examines all known European records of Purple Martin which are listed in Table 1.

Kingstown (Dun Laoghaire), Co Dublin, Ireland (autumn 1839 or 1840)

The Kingstown specimen of Purple Martin represents the first known record of the species in Europe. The female specimen was forwarded to Dr John Scouler for the purposes of dissection a few hours after being procured by Professor Frederick M'Coy, and was afterwards placed in the Museum of the Royal Dublin Society (RDS) (Yarrell 1840, 1843, 1876, Bond 1843, Thompson 1849, McGillivray 1852, Watters 1853, Harting 1866, Newman 1866, Gould 1873a, b, Dalgleish 1880). The specimen was subsequently transferred to the Museum of Science and Art around 1864 (O'Riordan 1983), and thence to the renamed National Museum of Ireland – Natural History [NMINH: 2003.52.1011 (DE-746)], where it is currently on public display (Figures 1, 2).

The date that the Kingstown specimen was apparently shot has variously been quoted as 'autumn 1839', 'sometime in 1839', 'probably in 1839', '1839', '1839 or 1840', 'a short time previous to March 1840', 'about 1840', and '1840' (Watters 1853, Dalgleish 1880, Freke 1880, Seebohm 1884, More 1885, 1890, Saunders 1889, Christy 1890, Praeger 1893, Harting 1901, Ussher 1908, Nichols 1924, Humphreys 1937, Kennedy *et al.* 1954, Alexander & Fitter 1955, Kennedy 1961, Ruttledge 1966, 1975, Fitzpatrick 2003, Parkin & Knox 2010, Coyle *et al.* 2007). Thompson (1849) and Ussher & Warren

Table 1. Details of Western Palearctic records of Purple Martin *Progne subis.*

Location	Date	Method	Sex	Recorders	Category
Near Kingstown (Dun Laoghaire), Co Dublin, Ireland, Lat./Long. 53.294 6.134 W.	1839-40	shot	female[1]	unknown	
Brent Reservoir, Kingsbury, Middlesex, Greater London, Britain, Lat./Long. 51.571 0.247 W.	First week Sept. 1842	shot	juvenile male of the year[2]	John Calvert	unaccepted
Brent Reservoir, Kingsbury, Middlesex, Greater London, Britain, Lat./Long. 51.571 0.247 W.	First week Sept. 1843	shot	old adult male[3]	John Calvert	unaccepted
West Colne Bridge, near Huddersfield, West Yorkshire, Britain, Lat./Long. 53.857 2.169 W.	1854	shot	unknown	unknown	unaccepted
Near Macclesfield, Britain, Cheshire, Lat./Long. 53.259 2.119 W.	Prior to 1861	shot	unknown[4]	Moses Armfield	unaccepted
River Stour, Wixoe Park, Suffolk, Britain, Lat./Long. 52.066 0.496 E.	c.1870	shot	unknown	John Squire	unaccepted
Colchester Barracks, Essex, Britain, Lat./Long. 51.896 0.892 E.	c.26 Sept. 1878	observed	unknown	Captain Dugmore	unaccepted
Pico, Azores, Portugal, Lat./Long. 38.458 28.323 W.	28 Sept. 1996	observed	2 juveniles	unknown	under review
Butt Lighthouse, Isle of Lewis, Outer Hebrides, Scotland, Britain, Lat./Long. 58.516 6.261 W.	5 Sept. 2004	observed	juvenile *et al.*	Shaun P. Coyle	A
Facho, Flores, Azores, Portugal, Lat./Long. 39.448 31.194 W.	6 Sept. 2004	observed	juvenile	Ingvar Torsson & Svante Aberg	A
Corvo, Azores, Portugal, Lat./Long. 39.673 31.115 W.	13-18 Oct. 2011	observed	juvenile	R. Mizrachi, R. Livne & B. Carlson	A

[1] NMINH:2003.52.1011 (DE-746).
[2] Booth Museum (Brighton).
[3] Formerly in John Calvert's Museum (London).
[4] Formerly in Armfield Museum (Macclesfield).

(1900) noted that Yarrell had received a letter from M'Coy about the specimen in March 1840. Initial details about the record were published the following month, in April 1840 (Yarrell 1840).

Although the identification of the extant Kingstown specimen as Purple Martin has never been questioned, the authenticity of this single Irish record has long been doubted. Ruttledge (1975) considered that 'though reliably identified, it is not yet fully admitted to the Irish List because the record may have been an escape from captivity or because it may not have reached the shore of Ireland alive'. Indeed, the record is currently included on the Irish List of Birds (IRBC, http://www.irbc.ie/) under Category D1, indicating uncertainty over its natural arrival.

While many authors did not appear to have any reservations regarding the authenticity of the Kingstown specimen (Yarrell 1840, 1843, 1876, Thompson 1849, McGillivray 1852, Watters 1853, Morris 1862, Harting 1866, 1901, Dalgleish 1880, Freke 1880, More 1885, 1890, Praeger 1893, Ussher & Warren 1900, Ussher 1908, Nichols 1924, Humphreys 1937, Alexander & Fitter 1955, Fitzpatrick 2003), others were politely sceptical, repeatedly stating that a specimen 'was said to have been shot near Kingstown' (Newman 1866, Gould 1873a, b, Seebohm 1884, Lilford 1897, Christy 1890, Kennedy *et al.* 1954, Kennedy 1961, Ruttledge 1966, 1975). Saunders (1889) was more forthright in his suspicion, suggesting that 'assisted passage' may have been involved, implying that the specimen may have been directly imported from North America. However, Alexander & Fitter (1955) pragmatically remarked that 'though most previous writers have sheltered behind "said to have been" shot in Ireland, no evidence is adduced to show that it might have been shot anywhere else'.

Praeger (1893) remarked 'The Purple Martin is one of those American birds whose occurrence in Ireland certainly cannot be attributed to escape from confinement; especially in 1840 (when a specimen was secured near Dublin), it was hardly likely that an attempt to transport such a purely insectivorous bird across the Atlantic could have been successful, as that was long before the days of ocean racers. That a bird of such great powers of flight and migratory habits should wander far from its native land is to be expected, and as this species is very abundant on the North American continent, it is not so surprising that wanderers should occasionally reach the western coasts of Europe'.

The fact that M'Coy sent the specimen to Scouler for the purposes of dissection a few hours after being procured, strongly suggests that it was freshly dead and had been shot locally. Although the person who shot the Kingstown specimen is unknown, the impressive professional credentials of both M'Coy and Scouler need to be taken into account in assessing the potential veracity of the record.

Sir Frederick M'Coy M.D., D.Sc., F.R.S. (1823-1899) was born in Dublin and studied medicine there and at Cambridge, but was drawn to natural science from an early age. He was appointed Professor of Geology at Queen's University, Belfast in 1852, but left two years later to assume the Chair of Natural History in the new University of Melbourne, where he was subsequently recognised as the leading man of science in Australia (Praeger 1949, Wyse-Jackson & Monaghan 1994).

Dr John Scouler M.D., L.L.D. (1804-1871) was a Scottish-born medical surgeon and accomplished naturalist. In 1833, he was appointed Professor of Minerology, and subsequently of geology, zoology, and botany, to the Royal Dublin Society (RDS), a post he held until his retirement in 1854, when he returned to Scotland (Praeger 1949, Nelson *et al.* 2014).

Kingsbury (Brent) Reservoir, Greater London, Middlesex, UK (September 1842)

Yarrell (1843, 1876) remarked: 'It is said that in the first week of September, 1842, two examples of this species (*P. subis*) were shot by John Calvert (1814-1897) of Paddington at the Brent (Kingsbury) Reservoir'. Yarrell (1843) noted that the two specimens had not been shot on the same day ('two to three days intervened'), and even suggested that a 'brood might therefore have been raised in this country'.

One of the specimens, a juvenile male of the year, which was examined by Yarrell (1843), passed into Mr Bond's collection (Bond 1843). Harting (1889) described Frederick Bond F.Z.S., F.E.S. (1811-1889) of Kingsbury as a noted English naturalist, possessing sufficient means to render him independent of a profession. Palmer (2000) noted that Bond's Purple Martin, which was included in a case (No 114) containing a Sand Martin *Riparia riparia,* a white Sand Martin, a Common Swift *Apus apus*, and an Alpine Swift *Tachymarptis melba,* was later purchased by Sir Vauncey Harpur Crewe (1846-1924) of Calke Abbey, near Ticknall, Derbyshire (Purcell & Thwaite 2013), and following his death in 1924, bought at auction for the Booth Museum, Brighton on 19 May 1925 for £7 10s (Frohawk 1925, Glegg 1935, Chalmers-Hunt 1976), where the specimen is still on public display. The second specimen, examined and described by Yarrell (1843) as an old adult male, was retained by Calvert. Calvert's extensive private museum collections in London were subsequently auctioned by J.C. Stevens during November 1897 and July 1898 (Stevens 1897, 1898, Sherborn 1940, Chalmers-Hunt 1976). According to Stevens (1897), Calvert was 'disposing of his collection owing to his declining health, and the unsafe condition of his Museum House through the excavations of the Midland Railway'. Calvert died in 1897 and the ultimate fate of his Purple Martin is unknown.

Palmer (2000) noted that although the Kingsbury records were initially accepted by some authorities, including Morris (1862), Newman (1866), Yarrell (1876) and Freke (1880), Dalgleish (1880) considered that Yarrell had been misinformed. Seebohm (1884) was also sceptical. Harting (1886, 1889, 1901) suggested that some degree of fraudulent behaviour had taken place and subsequently discovered that Calvert had bought American skins and relaxed them for mounting in Britain. In his otherwise genial catalogue of natural history collectors, Sherborn (1940) notoriously described John Calvert as a 'fraudster, traveller, self-proclaimed mining expert, and mineral collector'. However, Taylor (2016) noted that 'Sherborn seems to have attributed to John some of the doings of Albert (1872–1946), John's also unscrupulous and then still alive grandson - or son: the Calverts were never too clear about this'.

Colne Bridge, Huddersfield, West Yorkshire, UK (1854)
According to Hobkirk (1868), a Purple Martin was reported to have been shot at Colne Bridge near Huddersfield (West Yorkshire) in 1854. Although Gould (1873a, b) and Dalgleish (1880) subsequently referred to the record they did not make any comment about its authenticity. However, Clarke & Roebuck (1881) and Seebohm (1884) remarked that the record requires investigation and confirmation. Harting (1901) listed the record as 'doubtful', and Nelson (1907) and Alexander & Fitter (1955) concluded that 'it is not possible to investigate the circumstance, and the record is to be considered unreliable and unprovable'. The ultimate fate of the Colne Bridge specimen is unknown.

Macclesfield, Cheshire, UK (prior to 1861)
Dalgleish (1880) noted that a Purple Martin 'said to have been shot near Macclesfield, Cheshire, was sold, with other birds from Macclesfield Museum, in London 1861'. On 14 June 1861 the contents of Armfield Museum (Macclesfield) was auctioned by J.C. Stevens in London (Buckland 1861, Sherborn 1940, Chalmers-Hunt 1976). According to the pre-auction advertisements, the lots consisted, *inter alia,* of a variety of stuffed animals and birds, including lions, tigers, monkeys, and numerous birds, including a few very rare British birds shot by the late Mr Armfield, viz. Swallow-tailed Kite *Elanoides forficatus,* Purple Martin, and a 'new' Woodcock (Stevens 1861a, b). Harting (1901) specifically noted that the Purple Martin realised £1 8s.

Although Albert Calvert (1892) stated that some of the Armfield bird collection, which originally formed part of the Leverian Museum Collections, was purchased by his ancestor, John Calvert (see Kingsbury records), it is unclear whether or not the Leverian Collections contained any specimens of Purple Martin . Sir Ashton Lever (1729-1788) was the wealthy squire of Alkrington Hall near Manchester who had a passion for collecting all kinds of natural objects. The entire contents of Lever's private museum were initially sold by lottery during 1786 and subsequently by auction during 1806 (Chalmers-Hunt 1976). Harting (1901) considered that Armfield's Purple Martin was 'of doubtful authenticity', and its ultimate fate is unknown.

Moses Armfield (1787-1861) was a silk manufacturer and bird preserver who operated a private natural history museum in a block of buildings enclosed by Catherine Street, Great King Street, Pinfold Street, and Pierce Street in Macclesfield. Although Armfield had originally been declared bankrupt during April 1829 (Anon 1829), he was still officially listed as a silk manufacturer and museum proprietor during 1848 (Reid 2018). Following the decline of the Macclesfield silk industry during the 1860s, and Armfield's death in 1861, the museum was converted into a foundry and subsequently into a block of flats during the 1990s (Ron Thorn pers. comm.).

River Stour, Wixoe, Suffolk, UK (c.1870)
Christy (1890) remarked that 'Mr Fitch also writes me that he remembers seeing Mr John Squire, formerly of Wixoe Park, shoot one (Purple Martin) on the Stour about twenty years ago, but he does not know the present whereabouts of the specimen'. Mr Edward Arthur Fitch F.L.S., F.E.S. (1854-1912) of Maldon (Essex), was President of the Essex Field Club (Christy 1890).

Colchester Barracks, Essex, UK (c.26 September 1878)
Bree (1878) reported that 'Captain Dugmore, formerly of Colchester, told him that about September 26th 1878, he saw a specimen of the Purple Martin (as *Hirundo purpurea*) on the barrack exercising ground here. Having lived in Canada, where the bird is plentiful, and being within ten yards of the Swallow as it sailed past him, he had no doubt whatever about the bird being the one indicated. The strong prevalence of westerly winds for the last month may be expected to have blown over many birds. Colchester may now boast of being the locality where three very rare specimens of the Hirundinidae and Cypselidae have been procured'. Christy (1890) remarked that 'This record seems very unsatisfactory, but if so good a naturalist as Dr Bree gave credence to it, it is perhaps as well to follow him'.

Dr Charles Robert Bree M.D., F.L.S., F.Z.S. (1811-1886) was a physician at the Essex and Colchester Hospital, joint editor of the Naturalist, a staunch Whig and ardent politician (J.P. for Essex & Suffolk), and vehement opponent of Charles Darwin's theory of evolution. In 1859, he published the four-volume *The History of the Birds of Europe not observed in the British Isles*. The enlarged second edition, published in 1875-76, included a fifth volume.

Discussion

There are many officially accepted records of vagrant Nearctic birds occurring naturally, albeit irregularly, including several British and Irish records dating from the 19th century (Gatke 1860, Dalgleish 1880, Freke 1880, Praeger 1893, Alexander & Fitter 1955, Palmer 2000, Dempsey & O'Clery 2002, Parkin & Knox 2010, Harrop *et al.* 2013). The recently confirmed observation of Purple Martin on the Isle of Lewis (Scotland) and the Azores (Flores and Corvo) proves beyond doubt that this Nearctic hirundine can occur naturally. Indeed, two other species of Nearctic hirundines have been reliably recorded from Britain: Tree Swallow *Trachycineta bicolor* and American Cliff Swallow *Petrochelidon pyrrhonota,* in addition to one confirmed record of the American Cliff Swallow from Ireland (Parkin & Knox 2010). Many hirundine species are well noted for their ability and resilience in undertaking exceptionally long annual migrations (Turner & Rose 1989). Indeed, Coyle *et al.* (2007) noted that Purple Martin are particularly powerful fliers and suggested that this may enable them to cope better with initial displacement and even reorientate towards their New World wintering regions. However, Parkin & Knox (2010) considered that most vagrants are doomed; there is little evidence that long-distance vagrant juveniles ever make it back to their natal areas.

The vast majority of vagrant Nearctic land birds are recorded in Britain and Ireland during the autumn, particularly September and October, coinciding with the period of their natural southerly migrations, and often associated with particular wind conditions (Parkin & Knox 2010). Elkins (1979, 2008) noted that the fluctuating number of vagrant Nearctic land birds in Britain and Ireland is related to atmospheric variability across the North Atlantic and population trends in North America. He also noted that while tropical storms play an indirect role in initiating vagrancy in autumn, spring vagrants are unaffected by these and almost certainly include birds undertaking northward migration on the 'wrong side' of the Atlantic.

It is interesting to note that most of the European records of Purple Martin were recorded during the autumn, particularly during September, and that at least some appeared to be related to unusual meteorological conditions.

Although the specific date of the Irish Kingstown record is unknown, according to Parkin & Knox (2010), the specimen was most likely shot during the autumn of 1839. It is interesting to note that one of the worst hurricanes ever recorded in Ireland occurred on the 6-7 January 1839 (Shields & Fitzgerald 1989, Lamb & Frydendahl 1991). Thompson (1839) reported on the associated destruction of various birds and fish, specifically noting that many dead and dying European Storm Petrels *Hydrobates pelagicus* were subsequently found in several central and eastern parts of the country, including Cavan town (Co. Cavan), Mullingar (Co. Westmeath), Kells (Co. Meath), Brown Hill (Co. Calow), Wicklow town (Co. Wicklow), Saintfield (Co. Down), and Belfast (Co. Down). Perhaps the Kingstown Purple Martin's displacement and arrival in Ireland may have been associated with this extreme storm event? According to Coyle *et al.* (2004), Purple Martin is the earliest tropical-wintering migrant to reach the North American continent, arriving in the southernmost states by mid-January.

Although no unusual British weather events have been previously associated with the two Brent Reservoir records of Purple Martin during the first week of September in 1842, it is interesting to note that during early September 1842 a powerful western Atlantic storm known as Antje's Hurricane, tracked generally westward, yielding widespread destruction across the Bahamas, before finally striking northern Mexico on 8 September (Ludlum 1963). Perhaps this storm may have contributed to the displacement of southerly migrating Purple Martin across the Atlantic?

Due to the lack of specific dates for the Colne Bridge (1854), Macclesfield (prior to 1861), and River Stour (c.1870) records, no unusual weather events can be associated with them.

In relation to the Colchester individual of Purple Martin observed during late September 1878, Christy (1890) remarked that 'the strong prevalence of westerly winds for the last month may be expected to have blown over many birds (from North America)'. Elkins (1979, 2008) noted that the occurrence of vagrant Nearctic land birds in Britain and Ireland during the autumn is related to atmospheric variability across the North Atlantic, but not necessarily due to tropical storms.

Although no specific extreme weather events were attributed to the reported observation of Purple Martin on Pico (Azores) on 28 September 1996, at least four hurricanes and one major tropical storm were recorded in the western Atlantic during August and September 1996 (Pasch & Avila 1999). Coyle et al. (2007) suggested that the confirmed observations of Purple Martin on the Isle of Lewis and Flores during early September 2004 may have been associated with Hurricane Gaston on the eastern seaboard of the USA during late August 2004.

It is also possible that the confirmed observation of two juvenile Purple Martins on Corvo (Azores) between 13 and 18 October 2011 may have been linked to the 2011 Atlantic hurricane season which was described as the fourth most active season since record keeping began in 1851. Indeed, 2011 accounted for a total of 20 major tropical storms, seven of which became hurricanes. Nine of these storms, including six hurricanes, occurred between September and October (Anon 2018).

In conclusion, all of the Irish and British 19th century records of Purple Martin are currently unaccepted, mainly because the supporting historical evidence was considered to be either inadequate or, in some cases, possibly fraudulent. Although it is acknowledged that the evidence presented in this account could not possibly address all of criteria required in authenticating an official record, in the light of the recently confirmed records from the Azores and Scotland, a revaluation of some Irish and British 19th century records of Purple Martin is warranted.

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