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PORTUGUESE SOCIETY FOR THE STUDY OF BIRDS

Avenida Columbano Bordalo Pinheiro, 87, 3º Andar.

1070-062 Lisboa – Portugal

T. +351 213 220 430 | F. +351 213 220 439

spea@spea.pt | www.spea.com

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# Monitoring waterbird populations in the Tejo estuary, Portugal: report for the decade 2007-2016

Monitorização de aves aquáticas  
no estuário do Tejo, relatório  
da década 2007-2016

Pedro M. Lourenço<sup>1</sup>, Hany Alonso<sup>2</sup>,  
José A. Alves<sup>3,4</sup>, Ana T. Carvalho,  
Teresa Catry<sup>1</sup>, Hélder Costa,  
Joana S. Costa<sup>3</sup>, Maria P. Dias<sup>5,6</sup>,  
Vitor Encarnação<sup>7</sup>, Pedro Fernandes<sup>8</sup>,  
Ana I. Leal<sup>9,10</sup>, Ricardo C. Martins<sup>9,11</sup>,  
Filipe Moniz<sup>12</sup>, Sara Pardal<sup>13</sup>,  
Afonso Rocha<sup>13</sup> & Carlos D. Santos<sup>14,15</sup>



**1** Centro de Estudos do Ambiente e do Mar (CESAM)/Departamento de Biologia Animal, Faculdade de Ciências, Universidade de Lisboa, Campo Grande, 1749-016 Lisboa, Portugal. **2** Departamento de Biologia Animal, Faculdade de Ciências, Universidade de Lisboa, Campo Grande, 1749-016 Lisboa. **3** DBIO & CESAM - Centro de Estudos do Ambiente e do Mar, Universidade de Aveiro, Aveiro, Portugal. **4** South Iceland Research Centre, University of Iceland, Selfoss, IS-800, Iceland. **5** BirdLife International, The David Attenborough Building, Pembroke Street, Cambridge CB2 3QZ, UK. **6** MARE - Marine and Environmental Sciences Centre, ISPA - Instituto Universitário, Rua Jardim do Tabaco 34, 1149-041 Lisboa, Portugal. **7** Centro de Estudos de Migrações e Proteção de Aves (CEMPA), Instituto da Conservação da Natureza e das Florestas (ICNF), IP, Av. Combatentes da Grande Guerra 1, 2890-015 Alcochete, Portugal. **8** Bis 29, Rue Jabal Toubkal, apt.11, Agdal, Rabat, Morocco. **9** CEABN/InBIO, Instituto Superior de Agronomia, Universidade de Lisboa, Tapada da Ajuda, 1349-017 Lisboa, Portugal. **10** cE3c - Centro de Ecologia, Evolução e Alterações Ambientais, Departamento de Biologia Animal, Faculdade de Ciências, Universidade de Lisboa, 1749-016 Lisboa, Portugal. **11** CIBIO/InBIO - Centro de Investigação em Biodiversidade e Recursos Genéticos, Universidade do Porto, Campus Agrário de Vairão, 4485-601 Vairão, Portugal. **12** Av. Professor Bento de Jesus Caraça n.º25 - 3.ºB, 2725-032 Mem Martins, Portugal. **13** MARE - Marine and Environmental Sciences Centre, Department of Life Sciences, University of Coimbra, 3000-456 Coimbra, Portugal. **14** Núcleo de Teoria e Pesquisa do Comportamento, Universidade Federal do Pará, Rua Augusto Correa 01, Guamá, 66075-110 Belém, Brazil. **15** Department of Migration and Immuno-ecology, Max Planck Institute for Ornithology, Am Obstberg 1, 78315 Radolfzell, Germany.

\* **Corresponding author:** P.M. Lourenço, email: p.m.g.lourenco@gmail.com, tel: (+351)938468585

## ABSTRACT

The Tejo estuary is a key site for wintering and migratory waterbirds both at the national and international levels. Here we report the main findings of an ongoing monitoring programme of waterbirds in the main high tide roosts of the estuary. A decade of monthly counts (between 2007 and 2016) revealed peaks in waterbird abundance occurring between August and February, with monthly averages of over 28000 birds. Overall, our data highlight the importance of this wetland during winter and autumn migration, with lower but also relevant numbers during spring migration.

## Introduction

Wetlands, such as estuaries and other coastal areas, play a key role in the life cycles of millions of waterbirds as wintering, breeding and migratory stopover sites (Boere et al. 2006). Due to their high productivity and typical location at the interface between oceans and rivers, coastal wetlands also attract human settlement and economic development, often leading to conflicts between human activities and the conservation of natural values (Mee 2012). Land reclamation, water and sediment regulation, fisheries, hunting, pollution (from industrial, agricultural and domestic sources) and climate change, all exert varying pressures on waterbird populations worldwide (Delany et al. 2010). Consequently, the conservation and management of coastal wetlands is an important global issue. For an informed application of conservation efforts, knowledge of the current status of different waterbird populations, as well as their annual trends, is critical. Such information will not only indicate which species are facing harsher conditions and threats, but also, when sufficiently detailed, pinpoint the sites and habitats where conservation actions are most urgently needed.

The Tejo estuary, located on the central coast of Portugal, is one of the key sites for waterbirds along the East Atlantic Flyway (EAF), particularly for migratory shorebirds. Internationally important numbers (i.e. >1% flyway population) of some species winter here, such as pied avocet *Recurvirostra avosetta*, black-tailed godwit *Limosa limosa*, grey plover *Pluvialis squatarola*, dunlin *Calidris alpina*, ringed plover *Charadrius hiaticula* and bar-tailed godwit *Limosa lapponica*, as well as over 1% of the flyway breeding population of black-winged stilt *Himantopus himantopus* (Delany et al. 2009). The Tejo estuary is also an important staging site for shorebirds migrating between high latitude breeding areas and African wintering sites. In fact, most spe-

cies reach peak abundance during migratory periods (Alves et al. 2011, Catry et al. 2011), emphasizing the need to monitor bird populations not only in winter, but throughout the year.

Beside shorebirds, the Tejo estuary also holds relevant numbers of other waterbirds, such as gulls, egrets and herons, wildfowl and flamingos, highlighting the national and international importance of this wetland as a legacy of natural values for future generations. In order to improve monitoring of waterbirds populations in the Tejo estuary and provide detailed information on local population status and phenology, we started a monitoring programme in 2007 aimed at obtaining monthly waterbird counts in the main high tide roosting sites. The objective of this monitoring programme was also to detect any relevant changes occurring at roosting sites. Over the years, this effort provided accurate information on the relative importance of different roosts and documented in detail the importance of this wetland during wintering and migratory periods (Alves et al. 2010, 2011, 2012, Catry et al. 2011). After ten years of continuous monitoring, we now summarize the main findings in terms of population sizes, phenological patterns and relative importance of different roosts, and provide estimates of population trends during this decade for some of the most relevant populations wintering and breeding in the Tejo estuary.

## Methods

### Study area and waterbird counts

From January 2007 to December 2016, monthly counts were performed in nine key high tide roosts of the Tejo estuary (Fig. 1). The nine monitored sites cover the range of roosting conditions found by waterbirds in the Tejo: saltpans partially converted to shrimp production (Vasa Sacos, Ribeira das

**Figure 1** - Map of the Tejo estuary including all monitored high tide roosts (red), the bare intertidal areas (grey) and saltmarsh areas (green). The green line indicates the limits of the Tejo Estuary Nature Reserve (RNET) and the dashed blue line shows the limits of the Tejo Estuary Special Protection Area (SPA). The saltmarshes monitored by ICNF roughly correspond to all saltmarsh areas located within the green line.

**Figura 1** - Mapa do estuário do Tejo incluindo todos os refúgios de preia-mar monitorizados (vermelho), as zonas intertidais (cinzento) e os sapais (verde). A linha verde indica os limites da Reserva Natural do Estuário do Tejo (RNET) e a linha tracejada azul mostra os limites da Zona de Protecção Especial (ZPE) do estuário do Tejo. Os sapais monitorizados pelo ICNF correspondem aproximadamente aos sapais incluídos dentro da linha verde.



Enguias, Samouco, Alhos Vedros/Moita), active saltpans managed for waterbirds (Samouco), abandoned saltpans (Vale de Frades, Samouco, Sarilhos, Seixal and Alhos Vedros/Moita), and saltmarsh (Vasa-Sacos, Sarilhos, Alhos Vedros/Moita, Coina and Corroios). These conditions have remained

mostly unaltered during the ten year survey period. Only the northern roosts (Vasa Sacos, Vale de Frades, Ribeira das Enguias and Samouco) are included in the local protected areas (Tejo Estuary Nature Reserve and Tejo Estuary Special Protection Area (SPA); Fig. 1).

In each count, we registered the number of individuals of each species of waterbirds, covering all species of the following orders: Anseriformes, Podicipediformes, Phoenicopteriformes, Gruiformes, Gaviiformes, Ciconiiformes, Pelecaniformes, Suliformes and Charadriiformes. Although it was not a key goal, we also recorded the number of raptors (Accipitriformes and Falconiformes) flying over the roosts during monitoring. Counts took place in a period of c. 8 days around the spring tide peak for each month, typically with high water height reaching at least 3.3 m, guaranteeing that no intertidal areas remained available for birds during high tide (Rosa et al. 2006). Counts were preferentially undertaken between 2 h prior and 2 h after the peak of high tide, when bird movements are minimal. Occasionally, some counts extended beyond this period due to logistical problems. All counts were performed by very experienced observers, so we expect no observer bias among roosts. Three roosts were not monitored over the whole ten year period: monitoring in Coina only started in 2009, Ribeira das Enguias was not covered in 2016 due to observer unavailability, and Vasa-Sacos was not monitored in 2007 as the sites was not in use as a waterbird roost at the time due to shrimp farming activities. Otherwise, counting effort was similar across roosts, with a maximum of 98.3% of all possible counts performed in Alhos Vedros/Moita and Corroios, and a minimum of 88.3% of all possible counts performed in Ribeira das Enguias.

### Dealing with missing counts

During the 10 year period we carried out 991 counts in individual roosts; however, due to logistical reasons, we missed 89 monthly counts. Data on total numbers counted include all available counts, without any corrections for missing values (Table 1). Data

on the relative importance of different roosts use the average count for each site/month over the ten year period, therefore, sample sizes may differ among sites and months (range: 5-10 samples per site/month), but the averages are comparable. Data on phenological patterns also use the average of each month, but in this case for all roosts combined, so any given month when not all roosts were counted was excluded from the analysis (included data ranges 5-9 samples for each month). Details on how the method for analysing population trends deals with missing data are specified below.

### Grouping waterbird species according to their phenology

Data on the phenology of individual species provide a broad picture of the relevance of the Tejo estuary for waterbird populations along the annual cycle. In order to group species according to their phenology, we used the standardized (x-mean/SD) average count for each species in each month in a UPGMA (Unweighted Pair Group Method with Arithmetic Mean Algorithm) cluster analysis (Gauch 1982). This analysis was restricted to the 40 most abundant species (see Table 1).

### Population trends from 2007 to 2016

To analyse population trends, we used the TRIM-software (TRENds and Indices for Monitoring data; Gregory et al. 2005, Pannekoek & Van Strien 2005) through package 'rtrim' that allows implementing TRIM within the R statistical environment (Boogart et al. 2016). TRIM is a widely used free-ware program with an efficient implementation of log-linear Poisson regression models to analyse time series of count data (Gregory et al. 2005). The estimation method in TRIM uses generalized estimating equations (GEE; see Liang & Zeger 1986), thereby



taking into account serial correlation and over-dispersion from Poisson distribution. The models were run for each species, and the number of birds counted in each roost was used as the dependent variable. Before calculating population trends, data from the missing counts were estimated, based on a GEE model with roost identity, year, and the interactions between these two variables. We thus estimated the population numbers for the missing counts using the average numbers within the roost when it was counted, and the trends over the years observed in other roosts. Population trends were then calculated based on this dataset with both the observed and estimated counts. Using the computed annual indices and taking into account their uncertainty, population trends were expressed as ratios of the population present in 2016 compared to 2007. The estimates of the trends are expected to be normally distributed (Gregory et al. 2005, Pannekoek & Van Strien 2005). Mean yearly change rate estimates and confidence intervals were used to classify the trends per year in six categories: “strong increase”, “moderate increase”, “stable”, “moderate decrease”, “steep decrease” and “uncertain” (Soldaat et al. 2007).

The analysis of population trends was restricted to the 40 most abundant species (Table 1). Trends were derived from January counts, as the analysis of phenological patterns showed it was the winter month when numbers were more stable (see Appendix 1) and with a smaller chance of migratory movements that may affect the results. In addition, January counts are also used by Wetlands International to analyse global population trends in waterbird populations, so using this month guarantees a straightforward easier comparison of local and global trends. Still, to confirm the trends obtained for January alone, we performed similar analysis considering the count data from

December and February. Additionally, for species that use high tide roosts as breeding areas (black-winged stilt, Kentish plover *Charadrius alexandrinus* and little tern *Sterna albifrons*), we estimated trends for their breeding populations. In this case, we used May counts as the phenological patterns exhibited by these species (see Appendix 1 in the Supporting Information) suggest that numbers in this month are less likely to be affected by stochastic effects such as “late springs” or by the built up in numbers that takes place after the end of the breeding season. In the case of little tern, we could only obtain a breeding trend as this species winters in Africa.

### Estimating the proportion of waterbirds in the Tejo estuary that use non-monitored roosts

The present monitoring effort covered all the key high tide roosts in the Tejo estuary that are accessible from land. However, the northeast part of the estuary comprises extensive saltmarsh areas that are also used by waterbirds during high tide and can only be accessed by boat. Coverage of those remote areas was beyond the scope of the present monitoring effort, but these areas are regularly monitored by Instituto da Conservação da Natureza e das Florestas (ICNF). We must be cautious when using these data, as monitoring of those saltmarshes areas was often not carried out according to the schedule defined for the remaining counts. In any case, we used available count data from these saltmarshes, collected in the months of January, February and December from January 2012 to January 2015 to roughly estimate the “number of birds missed” during our monitoring effort. We only present these data for shorebirds and wildfowl, as we were unable to obtain similar data for the remaining waterbird groups.

Table 1 - Monthly averages ( $\pm$ SD) of all waterbird species identified at the Tejo estuary during the present monitoring programme between 2007 and 2016. Species are ordered in descending order of abundance.

Tabela 1 - Médias mensais ( $\pm$ DP) de todas as espécies de aves aquáticas detectadas no estuário do Tejo durante o programa de monitorização em curso, entre 2007 e 2016. As espécies estão ordenadas por abundância decrescente.

SPECIES	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
<i>Calidris alpina</i>	9732 $\pm$ 1930	9688 $\pm$ 3177	5647 $\pm$ 2437	9084 $\pm$ 6618	5265 $\pm$ 3169	476 $\pm$ 438	3203 $\pm$ 2464	7897 $\pm$ 4119	5813 $\pm$ 1872	7350 $\pm$ 1633	9976 $\pm$ 2528	8541 $\pm$ 2607
<i>Limosa limosa</i>	3090 $\pm$ 1399	2279 $\pm$ 1447	1042 $\pm$ 1238	584 $\pm$ 381	325 $\pm$ 240	1047 $\pm$ 1088	3014 $\pm$ 1814	4411 $\pm$ 2078	5386 $\pm$ 2291	5193 $\pm$ 2841	5543 $\pm$ 6298	5698 $\pm$ 5412
<i>Larus ridibundus</i>	3080 $\pm$ 1088	2681 $\pm$ 1247	1471 $\pm$ 787	679 $\pm$ 369	236 $\pm$ 150	783 $\pm$ 523	4856 $\pm$ 3207	4521 $\pm$ 2131	4211 $\pm$ 1889	2238 $\pm$ 1043	1591 $\pm$ 698	1989 $\pm$ 1178
<i>Pluvialis squatarola</i>	2564 $\pm$ 763	2650 $\pm$ 639	2079 $\pm$ 574	2114 $\pm$ 881	390 $\pm$ 325	214 $\pm$ 186	295 $\pm$ 228	1854 $\pm$ 1271	4158 $\pm$ 1574	3159 $\pm$ 1516	3011 $\pm$ 1262	2863 $\pm$ 954
<i>Larus fuscus</i>	2027 $\pm$ 878	1443 $\pm$ 481	1512 $\pm$ 575	513 $\pm$ 356	375 $\pm$ 598	156 $\pm$ 120	774 $\pm$ 775	3012 $\pm$ 1315	2939 $\pm$ 1483	1786 $\pm$ 575	2247 $\pm$ 1020	1881 $\pm$ 968
<i>Tringa totanus</i>	1269 $\pm$ 209	987 $\pm$ 303	3203 $\pm$ 273	35 $\pm$ 24	21 $\pm$ 22	187 $\pm$ 160	1759 $\pm$ 824	1987 $\pm$ 1019	1576 $\pm$ 511	1745 $\pm$ 565	1430 $\pm$ 393	1232 $\pm$ 336
<i>Phoenicopterus roseus</i>	1144 $\pm$ 489	1426 $\pm$ 906	1266 $\pm$ 1048	1091 $\pm$ 987	1076 $\pm$ 1609	728 $\pm$ 847	947 $\pm$ 646	1129 $\pm$ 681	1374 $\pm$ 690	1352 $\pm$ 842	1081 $\pm$ 923	636 $\pm$ 484
<i>Recurvirostra avosetta</i>	1974 $\pm$ 707	1717 $\pm$ 764	658 $\pm$ 539	144 $\pm$ 120	131 $\pm$ 114	80 $\pm$ 102	174 $\pm$ 318	29 $\pm$ 42	65 $\pm$ 53	324 $\pm$ 270	1130 $\pm$ 289	1699 $\pm$ 591
<i>Charadrius hiaticula</i>	578 $\pm$ 323	430 $\pm$ 392	303 $\pm$ 172	254 $\pm$ 108	158 $\pm$ 101	34 $\pm$ 31	53 $\pm$ 49	2103 $\pm$ 1093	1876 $\pm$ 1014	687 $\pm$ 287	662 $\pm$ 315	565 $\pm$ 382
<i>Himantopus himantopus</i>	426 $\pm$ 80	402 $\pm$ 145	572 $\pm$ 155	734 $\pm$ 211	568 $\pm$ 148	665 $\pm$ 205	1061 $\pm$ 518	804 $\pm$ 396	617 $\pm$ 180	440 $\pm$ 143	430 $\pm$ 117	339 $\pm$ 159
<i>Limosa lapponica</i>	462 $\pm$ 212	277 $\pm$ 310	64 $\pm$ 74	49 $\pm$ 82	9 $\pm$ 17	20 $\pm$ 31	83 $\pm$ 120	839 $\pm$ 603	1411 $\pm$ 682	935 $\pm$ 843	865 $\pm$ 638	332 $\pm$ 235
<i>Anas crecca</i>	1394 $\pm$ 1024	804 $\pm$ 809	332 $\pm$ 679	0.2 $\pm$ 1	0.2 $\pm$ 1	0.2 $\pm$ 1	0.1 $\pm$ 0.3	9 $\pm$ 27	10 $\pm$ 10	285 $\pm$ 504	755 $\pm$ 1567	932 $\pm$ 834
<i>Calidris canutus</i>	195 $\pm$ 132	240 $\pm$ 288	154 $\pm$ 247	228 $\pm$ 393	371 $\pm$ 447	38 $\pm$ 67	58 $\pm$ 67	397 $\pm$ 290	605 $\pm$ 550	527 $\pm$ 583	360 $\pm$ 409	308 $\pm$ 340
<i>Fulica atra</i>	425 $\pm$ 450	336 $\pm$ 429	390 $\pm$ 464	167 $\pm$ 270	176 $\pm$ 255	200 $\pm$ 232	132 $\pm$ 174	101 $\pm$ 135	150 $\pm$ 307	439 $\pm$ 622	378 $\pm$ 491	503 $\pm$ 603
<i>Anas platyrhynchos</i>	307 $\pm$ 211	444 $\pm$ 255	408 $\pm$ 241	350 $\pm$ 225	384 $\pm$ 278	126 $\pm$ 81	87 $\pm$ 94	334 $\pm$ 632	173 $\pm$ 205	175 $\pm$ 127	132 $\pm$ 88	164 $\pm$ 132
<i>Egretta garzetta</i>	152 $\pm$ 43	113 $\pm$ 34	141 $\pm$ 73	124 $\pm$ 35	194 $\pm$ 93	330 $\pm$ 206	461 $\pm$ 274	523 $\pm$ 291	432 $\pm$ 227	269 $\pm$ 133	170 $\pm$ 65	113 $\pm$ 52
<i>Anas clypeata</i>	538 $\pm$ 395	730 $\pm$ 523	384 $\pm$ 463	3 $\pm$ 3	1 $\pm$ 1	1 $\pm$ 1	0.4 $\pm$ 1	4 $\pm$ 8	44 $\pm$ 65	255 $\pm$ 276	434 $\pm$ 589	337 $\pm$ 328
<i>Numenius arquata</i>	353 $\pm$ 122	378 $\pm$ 192	73 $\pm$ 102	46 $\pm$ 48	17 $\pm$ 16	87 $\pm$ 40	265 $\pm$ 171	249 $\pm$ 134	257 $\pm$ 115	302 $\pm$ 122	278 $\pm$ 114	225 $\pm$ 116
<i>Calidris ferruginea</i>	54 $\pm$ 67	69 $\pm$ 70	102 $\pm$ 91	178 $\pm$ 183	152 $\pm$ 192	86 $\pm$ 81	105 $\pm$ 186	691 $\pm$ 610	370 $\pm$ 384	259 $\pm$ 302	196 $\pm$ 220	74 $\pm$ 180
<i>Charadrius alexandrinus</i>	230 $\pm$ 254	102 $\pm$ 91	107 $\pm$ 96	75 $\pm$ 51	90 $\pm$ 40	129 $\pm$ 84	287 $\pm$ 175	402 $\pm$ 192	323 $\pm$ 305	179 $\pm$ 162	206 $\pm$ 127	138 $\pm$ 86
<i>Arenaria interpres</i>	299 $\pm$ 106	179 $\pm$ 88	295 $\pm$ 189	243 $\pm$ 150	73 $\pm$ 102	46 $\pm$ 43	66 $\pm$ 48	167 $\pm$ 105	156 $\pm$ 75	229 $\pm$ 101	249 $\pm$ 101	209 $\pm$ 108
<i>Ardea cinerea</i>	136 $\pm$ 42	89 $\pm$ 30	61 $\pm$ 25	46 $\pm$ 14	51 $\pm$ 16	94 $\pm$ 67	110 $\pm$ 40	164 $\pm$ 55	160 $\pm$ 52	154 $\pm$ 92	155 $\pm$ 54	132 $\pm$ 62
<i>Phalacrocorax carbo</i>	233 $\pm$ 198	153 $\pm$ 72	73 $\pm$ 38	11 $\pm$ 10	1 $\pm$ 1	0.3 $\pm$ 1	19 $\pm$ 57	3 $\pm$ 6	25 $\pm$ 21	146 $\pm$ 137	300 $\pm$ 206	210 $\pm$ 142
<i>Tringa nebularia</i>	102 $\pm$ 26	119 $\pm$ 34	133 $\pm$ 38	130 $\pm$ 45	9 $\pm$ 8	11 $\pm$ 16	31 $\pm$ 19	123 $\pm$ 70	130 $\pm$ 66	131 $\pm$ 43	127 $\pm$ 54	93 $\pm$ 29
<i>Calidris minuta</i>	81 $\pm$ 68	106 $\pm$ 135	129 $\pm$ 95	144 $\pm$ 150	44 $\pm$ 62	0.1 $\pm$ 0.3	1 $\pm$ 2	76 $\pm$ 72	119 $\pm$ 74	136 $\pm$ 71	155 $\pm$ 130	58 $\pm$ 30

AIRO Waterbird monitoring in the Tejo estuary (2007-2016)

SPECIES	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
<i>Larus melanocephalus</i>	10±3	9±23	12 ±22	1±3	2±5	91 ±165	429 ±131	364 ±353	66±81	4±11	10±32	17±53
<i>Numenius phaeopus</i>	40±23	31±33	20±19	175 ±73	28±16	53±53	164 ±150	149 ±71	93±61	58±52	37±50	60 ±86
<i>Vanellus vanellus</i>	66±78	52±47	1±1	6±18	1±2	5±11	13 ±25	78 ±77	95±85	265 ±216	228 ±222	62 ±78
<i>Tringa erythropus</i>	58±47	43±28	67±38	56±36	1±1	5±9	22±21	163 ±298	78±46	75±46	67±51	58±58
<i>Platalea leucorodia</i>	91±33	78±42	55±45	24±24	22±19	25±32	30±58	18±19	55±50	119 ±89	80±24	84±60
<i>Calidris alba</i>	68 ±104	70±43	65±57	75±93	8±10	0.1 ±0.3	0.2 ±0.4	34±55	124 ±78	111 ±94	27±26	23 ±33
<i>Sternula albifrons</i>	0.0 ±0.0	0.3±1	0.0 ±0.0	70±49	89±46	105 ±59	137 ±99	121 ±120	15±27	0.2 ±0.4	0.0 ±0.0	0.0 ±0.0
<i>Plegadis falcinellus</i>	201 ±635	31±95	10±16	11±26	11±22	2±5	28±55	77 ±100	49 ±121	108 ±334	6±13	1±4
<i>Anas penelope</i>	90±85	156 ±231	3±6	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	1±2	73±88	105 ±173	76±90
<i>Actitis hypoleucos</i>	34±18	31±14	34±15	35±12	1±1	2±4	38±23	70±31	66±36	49±35	33±10	28±8
<i>Tachybaptus ruficollis</i>	56±42	29±29	37±85	4±6	2±3	8±13	18±18	45±48	37±34	57±49	56±61	61±53
<i>Philomachus pugnax</i>	5±4	12±17	64±84	22±16	0.4±1	0.2 ±0.4	15±11	151 ±259	75±42	36±39	2±2	4±7
<i>Haematopus ostralegus</i>	27±45	58±58	23±45	12±37	0.0 ±0.0	12±38	2±6	60±66	58±92	20±32	16±48	10±28
<i>Bubulcus ibis</i>	5±7	16±28	11±14	22±25	42±66	27±52	23±39	26±34	21±17	32±37	7±9	8±12
<i>Thalasseus sandvicensis</i>	10±7	10±12	12±9	4±7	0.2±1	13±41	10±22	49±47	59±44	27±13	23±15	14±13
<i>Gallinago gallinago</i>	11±12	18±15	18±29	1±1	0.1 ±0.3	2±7	0.0 ±0.0	1±2	96 ±293	5±5	9±13	9 ±10
<i>Anas acuta</i>	0.1 ±0.3	0.0 ±0.0	0.4±1	0.4±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	3±9	79 ±187	21±36	0±0.0
<i>Larus michabellis</i>	24±62	2±5	1±3	0.4±1	0.2 ±0.4	7±22	6±12	43±96	10±15	0.2±1	0.4±1	6±9
<i>Pluvialis apricaria</i>	39 ±112	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	16±44	43±95	1±1
<i>Gelochelidon nilotica</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	2±3	1±2	0.4±1	89 ±128	1±1	0.0 ±0.0	1±2	0.0 ±0.0	0.0 ±0.0
<i>Tadorna tadorna</i>	13±23	9±12	7±6	9±9	5±4	2±3	3±3	0.3±1	4±6	7±9	3±6	13±27
<i>Anas strepera</i>	8±14	6±10	11±7	8±10	15±29	4±7	1±2	0.2±1	1±2	1±1	3±8	5±14
<i>Tringa ochropus</i>	5±5	4±5	5±5	16±36	0.0 ±0.0	0.1 ±0.3	4±4	9±18	4±4	4±5	3±3	4±4
<i>Gallinula chloropus</i>	5±3	5±3	5±2	5±5	5±4	3±2	2±3	2±1	1±2	4±4	3±2	13±26
<i>Egretta alba</i>	4±4	2±2	3±4	2±2	2±3	1±2	1±1	1±1	2±4	3±3	4±5	5±9
<i>Anser anser</i>	7±14	13 ±39	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	1±4	0.4±1	0.0 ±0.0
<i>Ciconia ciconia</i>	0.3±1	0.1 ±0.3	0.0 ±0.0	0.2±1	0.2±1	0.0 ±0.0	0.3 ±0.4	13±61	6±24	0.0 ±0.0	0.0 ±0.0	0.2±1
<i>Netta rufina</i>	0.0 ±0.0	1±3	2±3	4±8	4±7	1±2	0.0 ±0.0	0.2±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Glareola pratincola</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	2±6	2±4	7±16	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0

AIRO Monitorização de aves aquáticas no estuário do Tejo (2007-2016)

SPECIES	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
<i>Chlidonias niger</i>	0.0 ±0.0	0.0 ±0.0	0.3±1	1±2	3±5	1±2	0.2 ±0.4	1±1	2±6	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0
<i>Larus minutus</i>	0.0 ±0.0	8±24	0.2 ±0.4	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3
<i>Mergus serrator</i>	2±5	2±6	2±3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.4±1	1±2
<i>Sterna hirundo</i>	0.0 ±0.0	1 ±3	0.0 ±0.0	0.3 ±1	0.0 ±0.0	1 ±2	0.4 ±1	1 ±1	1 ±2	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Charadrius dubius</i>	0.0 ±0.0	0.0 ±0.0	0.4 ±1	0.2 ±0.4	0.0 ±0.0	0.1 ±0.3	1 ±2	2 ±6	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Ardea purpurea</i>	0.0 ±0.0	0.0 ±0.0	0.2 ±0.4	1 ±1	1 ±2	1 ±2	0.3 ±0.4	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Burhinus oedicanus</i>	0.0 ±0.0	0.1 ±0.3	0.2 ±1	0.3 ±1	0.1 ±0.3	2 ±4	0.0 ±0.0	0.1 ±0.3	1 ±2	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0
<i>Aythya fuligula</i>	1±3	0.2±1	0.2±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	2±6	0.0 ±0.0
<i>Hydroprogne caspia</i>	0.1 ±0.3	1±1	0.4±1	0.3 ±0.4	0.0 ±0.0	0.2±1	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.4±1	0.0 ±0.0	1±2
<i>Podiceps nigricollis</i>	0.1 ±0.3	0.3±1	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	1±3	1±2
<i>Chlidonias hybridus</i>	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.1 ±0.3	2±5	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Tringa glareola</i>	0.3±1	0.1 ±0.3	1±2	0.2 ±0.4	0.0 ±0.0	0.0 ±0.0	0.2±1	0.4±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Alopochen aegyptiaca</i>	0.0 ±0.0	0.2 ±0.4	0.2±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.3±1	0.0 ±0.0	0.3±1	0.0 ±0.0
<i>Marmaronetta angustirostris</i>	0.0 ±0.0	0.0 ±0.0	0.3±1	0.3±1	0.1 ±0.3	0.3±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Porphyrio porphyrio</i>	0.0 ±0.0	0.2±1	0.0 ±0.0	0.3±1	0.3±1	0.2±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Egretta gullaris</i>	0.0 ±0.0	0.0 ±0.0	0.2 ±0.4	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.3±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Tadorna ferruginea</i>	0.0 ±0.0	0.4±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0
<i>Aythya ferina</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.3±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3
<i>Limnodromus scolopaceus</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0
<i>Phalaropus fulicarius</i>	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.2 ±0.4	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0
<i>Tringa stagnatilis</i>	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.2 ±0.4	0.0 ±0.0
<i>Ardeola ralloides</i>	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Calidris temminckii</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.2±1
<i>Cygnus olor</i>	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Larus genei</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.3±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Podiceps cristatus</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.2 ±1
<i>Rallus aquaticus</i>	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0
<i>Branta canadensis</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Fulica cristata</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.2 ±0.4	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0

SPECIES	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
<i>Gavia immer</i>	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0
<i>Ixobrychus minutus</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.2±1	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Larus canus</i>	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Phalaropus lobatus</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Rissa tridactyla</i>	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3
<i>Tringa flavipes</i>	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Tringa semipalmata</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Clangula hyemalis</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0
<i>Larus delawarensis</i>	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Larus hyperboreus</i>	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0
<i>Melanitta fusca</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0
<i>Phoeniconaias minor</i>	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.3	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0

## Results

### Waterbird community: composition and abundance

We identified a total of 95 waterbird species during roost counts (Table 1), the most common groups being shorebirds (36 species), followed by wildfowl (19 species), gulls and terns (17 species), and egrets and herons (8 species). We also identified five rallids, three grebes, two flamingos, one ibis, one spoonbill, one cormorant, one stork and one loon (Table 1). This list includes six species that are considered vagrants in the Western Palearctic, two originating from Africa (reef egret *Egretta gullaris* and lesser flamingo *Phoeniconaias minor*), and four from North America (lesser yellowlegs *Tringa flavipes*, willet *Tringa semipalmata*, long-billed dowitcher *Limnodromus scolopaceus* and ring-billed gull *Larus delawarensis*). Another four species are most likely escapes from captivity (Canada goose *Branta canadensis*, mute swan *Cygnus olor*, ruddy shelduck *Tadorna ferruginea* and Egyptian goose *Alopochen aegypt-*

*ticus*) and we cannot rule-out the possibility that this was also the case of lesser flamingo as this species is often kept in captivity.

Over the ten years of monitoring we counted nearly three million birds, the ten most abundant species being dunlin, black-tailed godwit, black-headed gull *Larus ridibundus*, grey plover, lesser black-backed gull *Larus fuscus*, greater flamingo *Phoenicopterus roseus*, common redshank *Tringa totanus*, pied avocet, common ringed plover *Charadrius hiaticula* and black-winged stilt (Table 1). Among the 40 most abundant species, which were analysed in greater detail, we mainly found shorebirds (22 species), gulls and terns (five species), ducks (four species) and herons and egrets (three species).

During the counts, we also detected 14 raptor species, the most abundant of which were marsh harrier *Circus aeruginosus*, common kestrel *Falco tinnunculus*, osprey *Pandion haliaetus*, Eurasian buzzard *Buteo buteo* and black-shouldered kite *Elanus caeruleus* (Table 2).

**Table 2** - Monthly averages ( $\pm$ SD) of raptor species identified at the Tejo estuary during the present monitoring programme between 2007 and 2016. Species are ordered in descending order of abundance.

**Tabela 2** - Médias mensais ( $\pm$ DP) de aves de rapina detectadas no estuário do Tejo durante o programa de monitorização em curso, entre 2007 e 2016. As espécies estão ordenadas por abundância decrescente.

SPECIES	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
<i>Circus aeruginosus</i>	4 $\pm$ 4	3 $\pm$ 3	4 $\pm$ 4	4 $\pm$ 4	4 $\pm$ 3	3 $\pm$ 3	4 $\pm$ 3	4 $\pm$ 3	2 $\pm$ 2	4 $\pm$ 3	4 $\pm$ 3	5 $\pm$ 4
<i>Falco tinnunculus</i>	2 $\pm$ 2	2 $\pm$ 2	1 $\pm$ 2	0.4 $\pm$ 1	0.3 $\pm$ 0.4	1 $\pm$ 1	2 $\pm$ 1	1 $\pm$ 1	0.3 $\pm$ 1	2 $\pm$ 2	2 $\pm$ 2	2 $\pm$ 2
<i>Pandion haliaetus</i>	2 $\pm$ 1	1 $\pm$ 1	1 $\pm$ 1	0.4 $\pm$ 1	1 $\pm$ 1	0.2 $\pm$ 0.4	0.3 $\pm$ 1	1 $\pm$ 1	2 $\pm$ 1	2 $\pm$ 1	2 $\pm$ 2	2 $\pm$ 1
<i>Buteo buteo</i>	0.3 $\pm$ 1	1 $\pm$ 1	0.3 $\pm$ 1	0.3 $\pm$ 1	0.3 $\pm$ 1	1 $\pm$ 1	1 $\pm$ 1	0.1 $\pm$ 0.3	7 $\pm$ 20	1 $\pm$ 1	1 $\pm$ 1	1 $\pm$ 1
<i>Elanus caeruleus</i>	2 $\pm$ 2	1 $\pm$ 2	1 $\pm$ 1	0.4 $\pm$ 1	0.3 $\pm$ 1	1 $\pm$ 3	0.0 $\pm$ 0.0	2 $\pm$ 2	1 $\pm$ 2	2 $\pm$ 2	2 $\pm$ 2	2 $\pm$ 2
<i>Falco peregrines</i>	1 $\pm$ 1	1 $\pm$ 1	0.3 $\pm$ 1	0.4 $\pm$ 1	0.2 $\pm$ 0.4	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.2 $\pm$ 1	0.4 $\pm$ 1	1 $\pm$ 1	0.4 $\pm$ 1	0.3 $\pm$ 1
<i>Milvus migrans</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.2 $\pm$ 0.4	0.3 $\pm$ 1	0.3 $\pm$ 1	1 $\pm$ 1	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0
<i>Circus cyaneus</i>	0.2 $\pm$ 0.4	0.2 $\pm$ 0.4	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.3 $\pm$ 1	0.3 $\pm$ 1	0.1 $\pm$ 0.3
<i>Aquila fasciata</i>	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.2 $\pm$ 0.4	0.1 $\pm$ 0.3	0.2 $\pm$ 0.4	0.1 $\pm$ 0.3
<i>Aquila pennata</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.2 $\pm$ 1	0.2 $\pm$ 0.4	0.1 $\pm$ 0.3	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.3
<i>Milvus milvus</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0
<i>Circaetus gallicus</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0
<i>Circus pygargus</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0
<i>Falco columbarius</i>	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.1 $\pm$ 0.3

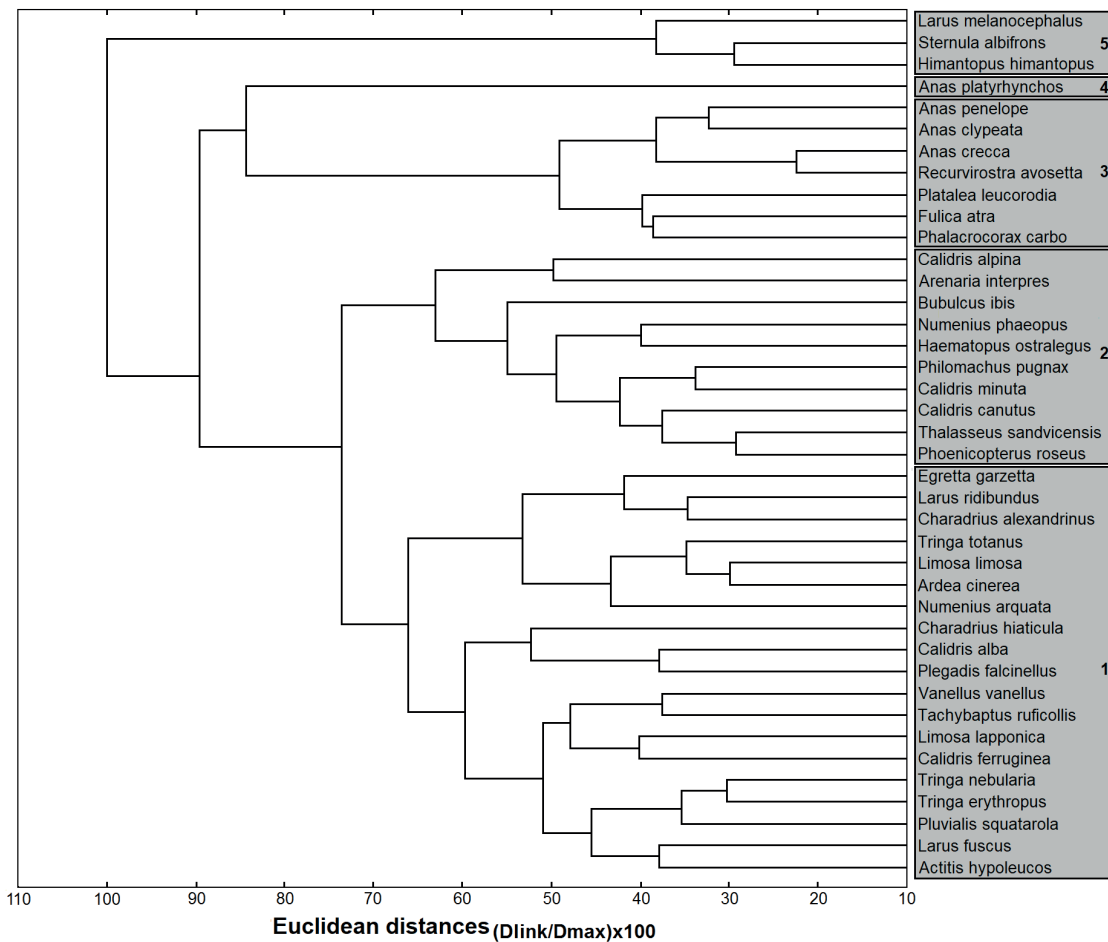
### Phenological patterns

Overall, waterbird abundances in the monitored roosts peaked between August and February, when the average number of birds reached 28500-33500. By March-April numbers had dropped to roughly 18000 by March-April, reaching their lowest point in June when total numbers were on average below 6000 birds. This evidences that the vast majority of species rely on the Tejo estuary during wintering and migratory passage periods (Fig. 2; see Appendix 1 in the Supporting Information for the detailed phenological patterns of all the 40 most abun-

dant species). Despite considerable variation among species, there were peaks associated with autumn migration for gulls and terns, occurring in July-September, and for flamingos, herons and egrets, mostly during August-October. Wildfowl did not show any evident migratory peaks, being mostly present in mid-winter from December to February (Fig. 2B). Shorebird numbers were high from August through February, with some species showing clear increases in numbers during autumn migration, others peaking during winter, and some species with a smaller but discernible peak in April associated with spring migration.

Figure 2 - Dendrogram based on Euclidean distances, representing the similarities among waterbird species with respect to phenological pattern.

Figura 2 - Dendograma baseado em distâncias euclidianas que representa as semelhanças entre espécies de aves aquáticas no que respeita aos seus padrões fenológicos.



A cluster analysis allowed the identification of 5 groups of species in terms of their phenology in the Tejo estuary (Fig. 3). The larger group included 19 species, such as black-tailed godwit, black-headed gull, grey plover and lesser black-backed gull, for which counts showed a clear increase in numbers during autumn migration and in most cases remained high during mid-winter. The second group included 10 species, such as dunlin, greater flamingo and red knot *Calidris canutus*, which generally showed increases

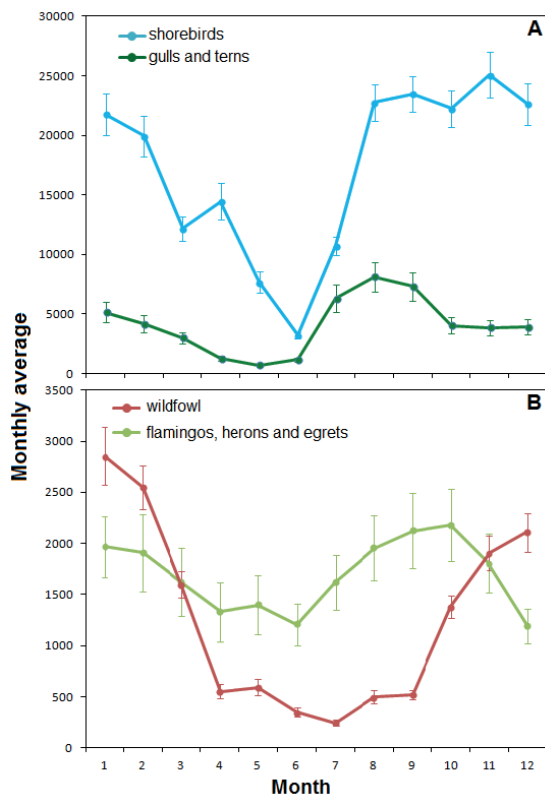
during spring migration or had peaks during both migratory periods, while remaining abundant in mid-winter. The third group was defined by a clear peak in mid-winter, generally between November and February, and included species such as pied avocet, great cormorant *Phalacrocorax carbo*, Eurasian coot *Fulica atra* and several ducks (*Anas crecca*, *A. clypeata* and *A. penelope*).

A fourth group included only mallard *Anas platyrhynchos* which had a particular phenological pattern with lower numbers

in June-November, an increase over December and January, and a long peak in February-May (see Appendix 1 in the Supporting Information). Finally, the fifth group included the three species that peak in late spring and early summer (generally between May and August), including black-winged stilt, little tern and Mediterranean gull *Larus melanocephalus* (Fig. 3).

**Figure 3** - Monthly average ( $\pm$ SD) of waterbirds counted in Tejo estuary's high tide roosts for the period 2007-2016. Panel A: shorebirds and gulls and terns. Panel B: wildfowl and flamingos, herons and egrets.

**Figura 3** - Média mensal ( $\pm$ DP) de aves aquáticas contadas nos refúgios de preia-mar do estuário do Tejo no período 2007-2016. Paineira A: aves limícolas (azul) e gaivotas e andorinhas do mar (verde escuro). Paineira B: anatídeos (vermelho) e flamingos e garças (verde claro).



### Relative importance of different roosts

Overall, during the ten years of monitoring, the most important high tide roost for waterbirds in the Tejo estuary was Samouco,

receiving on average over 30% of all counted birds (Table 3). Together with Vasa Sacos (22.4%), Ribeira das Enguias (12.3%) Corroios (10.6%) and Alhos Vedros/Moita (7.9%), these five roosting sites harboured over 85% of all birds present during our counts (Table 3). The importance of these five sites is similar for all waterbird groups. Together, these five roosts harboured 86% of shorebirds, 81% of all gulls and terns, 85% of all herons, egrets, flamingos and spoonbills (although in this case Sarilhos ranked as the fourth main roost with 10.3%) and 91% of all wildfowl (but Corroios was not particularly important for wildfowl with just 0.4%; Table 3).

These five main sites also remained the most significant throughout the annual cycle, although Samouco seems to be particularly important during autumn migration, while Vasa Sacos loses some of its relative importance during spring migration (Fig. 4). At the species level, Vasa Sacos is particularly important for black-tailed godwits during July-December, while Samouco, Ribeira das Enguias and Alhos Vedros are especially important for common redshank during autumn migration (roughly July-October; Fig. 4). Sarilhos, Samouco, Corroios and Alhos Vedros are particularly important for pied avocet in November-February. Vasa Sacos, Samouco and Ribeira das Enguias are very important for common ringed plover during autumn migration (August-September), and Samouco is virtually the only roost used by bar-tailed godwits during autumn migration (roughly August-November; Fig. 4). Samouco and Ribeira das Enguias are the two key sites for black-winged stilt throughout the year, but especially so after the breeding season in July, when the species peaks in the estuary (Fig. 4). The sites located within the Tejo SPA harbour most birds throughout the annual cycle, the main exception being pied avocet for which roosts outside the protected area receive most individuals between November and February, the period when this species peaks in the estuary.



Data from January suggest there are some temporal trends to the relative importance of difference roosts (Fig. 5). Overall, Vasa Sacos became more important over the years, in fact becoming the most important roost since 2014, while Samouco had much more birds in the first two years of monitoring after which it showed a decline and the numbers stabilized around 6000. Among the remaining roosts, numbers seem to fluctuate over the years without obvious trends, although both Alhos Vedros/Moita and Seixal seem to lose some importance over the years (Fig. 5).

Looking at different phenological and taxonomic groups (Fig. 6), the monthly averages over the years mostly followed the same trends as we observed in overall January counts, although the decline in Samouco seems to be mainly driven by shorebirds and gulls. For wildfowl, which form the bulk of phenological group 3, numbers fluctuated without a clear trend. The increase observed in Vasa Sacos is mostly explained by changes in the number of shorebirds and wildfowl, the same applying to phenological groups 1, 2 and 3 (Fig. 6)

**Table 3** - Relative importance of the different high tide roosts used by waterbirds at the Tejo estuary. For each roost we present monthly averages over the ten year period as well as the average percentage ( $\pm$ SD) of individuals counted in that particular roost (%). Data is presented for all birds and for each of four main groups: wildfowl (including Anatidae and Rallidae), flamingos, herons and egrets (including all Phoenicopteridae and Ardeidae), gulls and terns (including all Laridae and Sternidae) and shorebirds (including all Charadriidae, Scolopacidae, Recurvirostridae, Haematopodidae and Glareolidae).

**Tabela 3** - Importância relativa dos diferentes refúgios de praia-mar usados pelas aves aquáticas no estuário do Tejo. Para cada refúgio são apresentadas as médias mensais ao longo dos dez anos de monitorização, assim como a percentagem média ( $\pm$ DP) de indivíduos contados em cada refúgio (%). Os dados são apresentados para o conjunto de todas as aves aquáticas e para os quatro principais grupos.

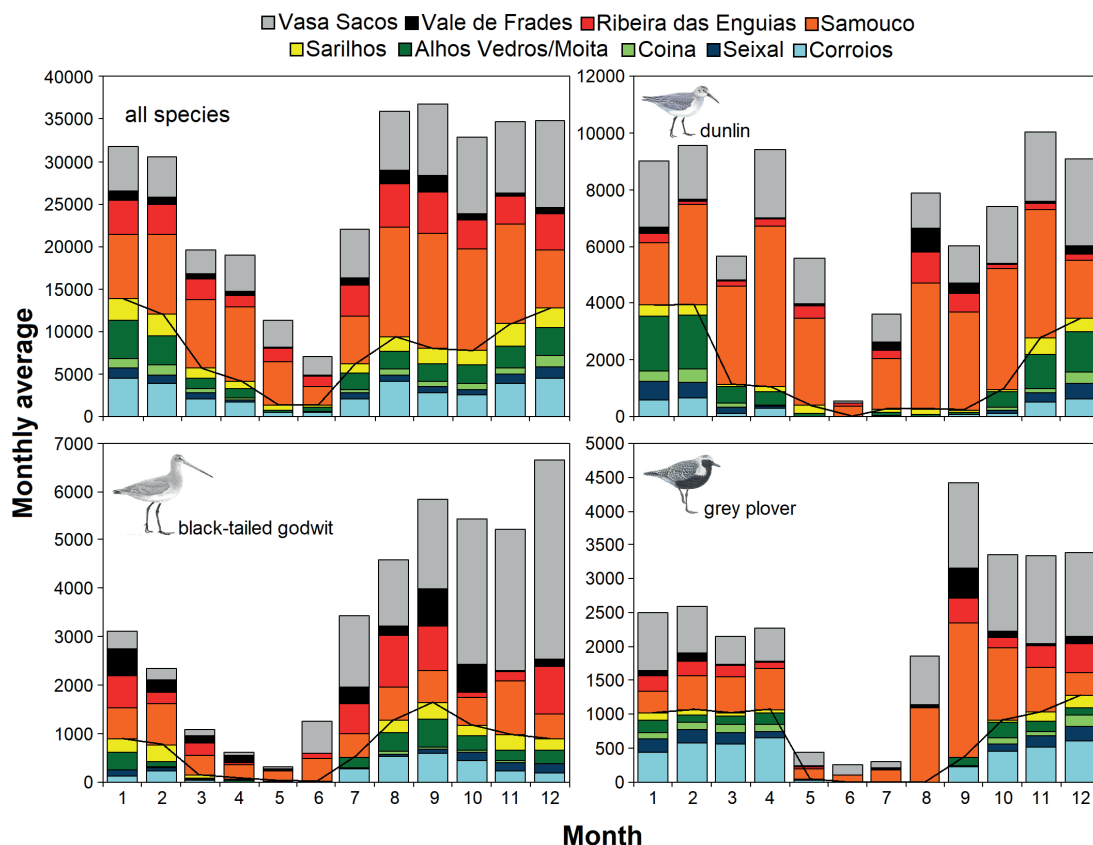
ROOST	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	%
ALL BIRDS													
Corroios	4503	4207	2140	1671	454	441	2090	4153	2862	2602	3943	4539	10.6 $\pm$ 3.1
Seixal	1222	1034	651	242	20	156	671	698	672	616	1064	1295	2.6 $\pm$ 1.1
Coina	1092	1406	533	273	27	43	410	747	616	736	733	1318	2.5 $\pm$ 1.3
Alhos Vedros/ Moita	4493	3387	1215	1144	272	420	1929	2093	2042	2104	2601	3354	7.9 $\pm$ 3.1
Sarilhos	2585	2487	1133	815	592	280	1093	1630	1799	1795	2605	2321	6.1 $\pm$ 1.5
Samouco	7489	9054	8059	8738	5035	2168	5593	12906	13536	11914	11725	6715	32.5 $\pm$ 8.2
Ribeira das Enguias	4059	3536	2463	1417	1604	1229	3636	5185	4890	3336	3255	4309	12.3 $\pm$ 2.8
Vale de Frades	1131	855	563	410	199	116	942	1593	1908	801	378	728	3.0 $\pm$ 1.3
Vasa Sacos	5250	4764	2848	4251	3182	2168	5699	6899	8384	8933	8335	10192	22.4 $\pm$ 5.6
SHOREBIRDS													
Corroios	2284	2607	1006	1082	92	182	749	1159	1383	1599	1995	2454	7.4 $\pm$ 2.9
Seixal	1191	983	558	204	14	6	82	129	188	497	1019	1271	2.8 $\pm$ 1.1
Coina	910	1096	413	182	11	4	202	250	277	487	545	1150	2.5 $\pm$ 1.7

ROOST	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	%
SHOREBIRDS													
Alhos Vedros/ Moita	3356	2553	881	901	168	150	991	1154	1315	1574	2254	2701	8.1 ±3.7
Sarilhos	1645	1449	368	424	354	119	451	783	679	736	1832	1826	4.8 ±1.9
Samouco	5171	6985	6180	8075	4516	1562	3956	10464	10855	9490	9510	4704	36.5 ±10.9
Ribeira das Enguias	1913	1096	1099	699	696	408	1759	3553	3015	1105	1517	2390	8.6 ±3.4
Vale de Frades	1049	740	444	268	129	22	656	1549	1822	668	309	642	3.7 ±2.1
Vasa Sacos	4758	3746	1997	3613	2340	1327	3718	5565	6122	7806	7375	8989	25.7 ±6.4
GULLS AND TERNS													
Corroios	2011	1664	1023	534	324	203	1235	2733	1247	776	1727	1934	26.9 ±10.6
Seixal	20	51	95	48	13	265	593	562	469	124	21	15	4.0 ±4.6
Coina	49	97	133	115	29	58	294	484	332	161	59	44	3.2 ±1.9
Alhos Vedros/ Moita	1033	846	300	214	99	243	903	1000	795	449	254	548	11.7 ±3.6
Sarilhos	639	705	533	245	116	100	559	645	738	655	473	302	10.0 ±3.5
Samouco	1225	954	956	141	75	161	1066	1823	1951	1644	1395	1076	21.7 ±8.5
Ribeira das Enguias	402	774	312	166	187	276	925	761	1034	664	254	360	10.7 ±4.5
Vale de Frades	3	8	113	51	9	8	256	371	30	133	60	4	1.8 ±1.5
Vasa Sacos	71	20	22	53	29	307	1656	984	1818	341	60	405	10.1 ±8.3
WILDFOWL													
Corroios	4	3	7	7	5	8	6	2	7	6	6	4	0.4 ±0.7
Seixal	1	1	2	3	2	1	0	0	0	1	0	1	0.1 ±0.2
Coina	90	174	3	2	0	1	2	2	1	63	100	100	3.5 ±2.5
Alhos Vedros/ Moita	5	4	7	8	7	5	1	0	2	4	9	11	0.4 ±0.4
Sarilhos	103	67	53	54	47	6	3	5	7	17	38	29	2.8 ±2.6
Samouco	537	708	424	192	174	95	31	36	50	150	234	329	19.5 ±8.3
Ribeira das Enguias	882	967	759	266	385	224	211	145	269	793	758	1215	45.2 ±11.4
Vale de Frades	57	56	23	42	16	9	10	7	8	21	26	70	2.3 ±1.5
Vasa Sacos	774	461	322	22	16	33	7	68	141	473	615	1006	25.9 ±12.9

ROOST	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	%
FLAMINGOS, HERONS AND EGRETS													
Corroios	202	117	104	48	33	48	99	258	224	219	213	145	7.7 ±3.4
Seixal	14	10	5	2	3	4	6	7	14	8	26	11	0.5 ±0.3
Coina	4	3	2	3	4	6	10	9	6	5	4	6	0.3 ±0.1
Alhos Vedros/ Moita	202	193	101	106	28	29	53	56	117	186	130	164	6.1 ±3.3
Sarilhos	195	264	179	91	74	44	80	197	373	383	261	157	10.3 ±4.4
Samouco	536	492	587	341	276	351	523	570	658	609	557	543	27.2 ±5.4
Ribeira das Enguias	704	465	256	278	356	347	725	688	578	711	435	279	26.2 ±6.8
Vale de Frades	28	55	73	89	67	90	118	48	75	93	41	13	3.6 ±1.9
Vasa Sacos	134	339	348	504	712	476	318	212	253	299	376	81	18.2 ±12.5

Figure 4 - Monthly average of waterbirds counted in each high tide roost of the Tejo estuary between 2007 and 2016 for all species combined (top left) and for the nine most abundant species (excluding gulls). Black lines represent the sums for the roosts laying in the southern part of the estuary, which are outside protected areas (see also Figure 1).

Figura 4 - Média mensal de aves aquáticas contadas em cada um dos refúgios de preia-mar do estuário do Tejo entre 2007 e 2016 para todas as espécies em conjunto (canto superior esquerdo) e para cada uma das nove espécies mais abundantes (excluindo gaivotas). As linhas pretas representam a soma de aves contadas nos refúgios da parte sul do estuário, que estão fora dos limites das áreas protegidas (ver também a Figura 1).



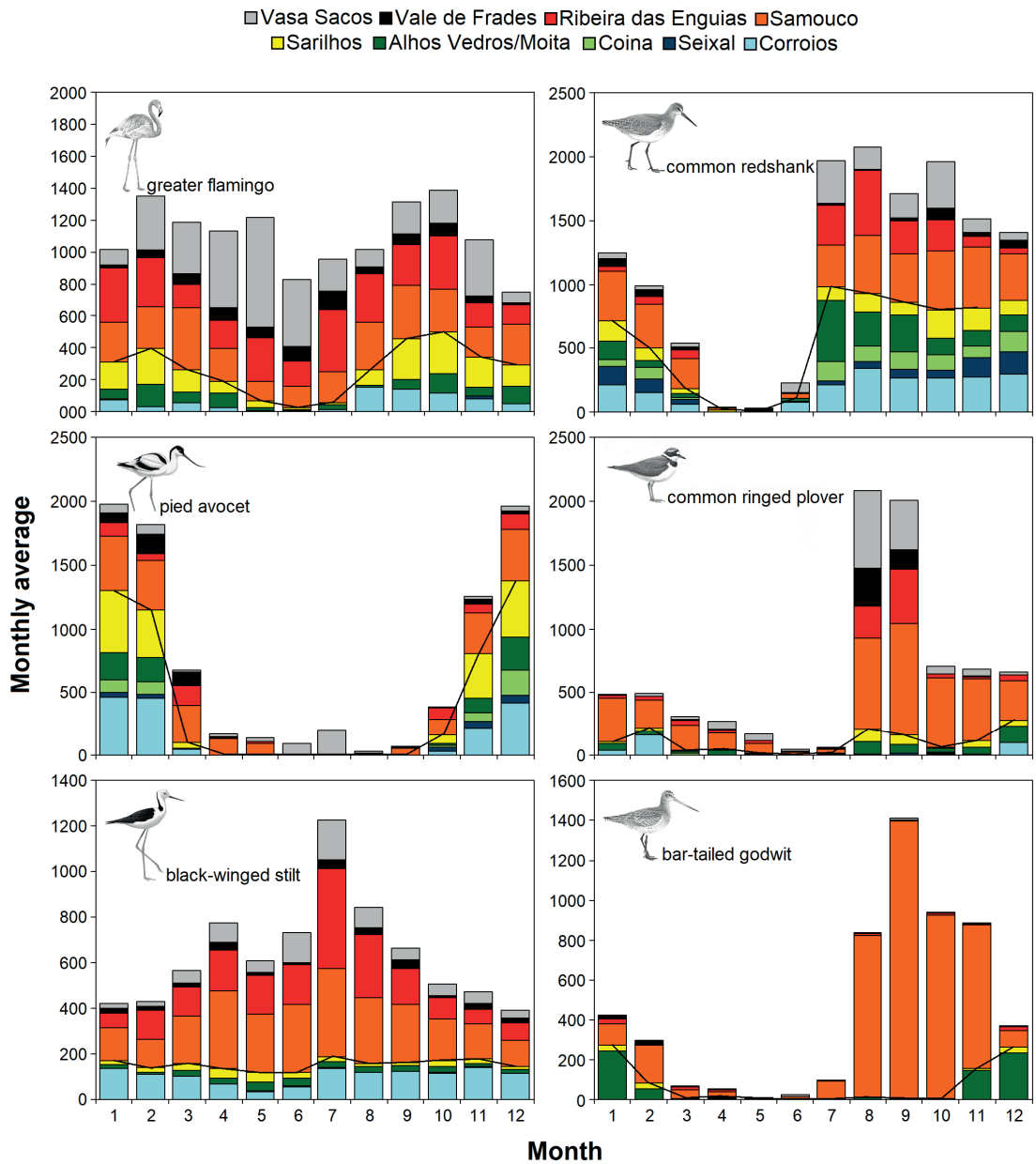


Figure 5 - Total waterbirds (all species combined) recorded in the January count of each high tide roost of the Tejo estuary between 2007 and 2016, showing changes in the relative importance of each roost over the ten year period.

Figura 5 - Total de aves aquáticas (todas as espécies em conjunto) contadas em Janeiro em cada um dos refúgios de preia-mar do Tejo entre 2007 e 2016, mostrando as mudanças na importância relativa dos diferentes refúgios ao longo destes dez anos.

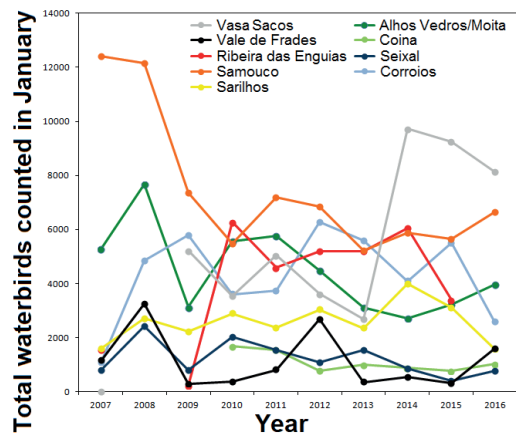
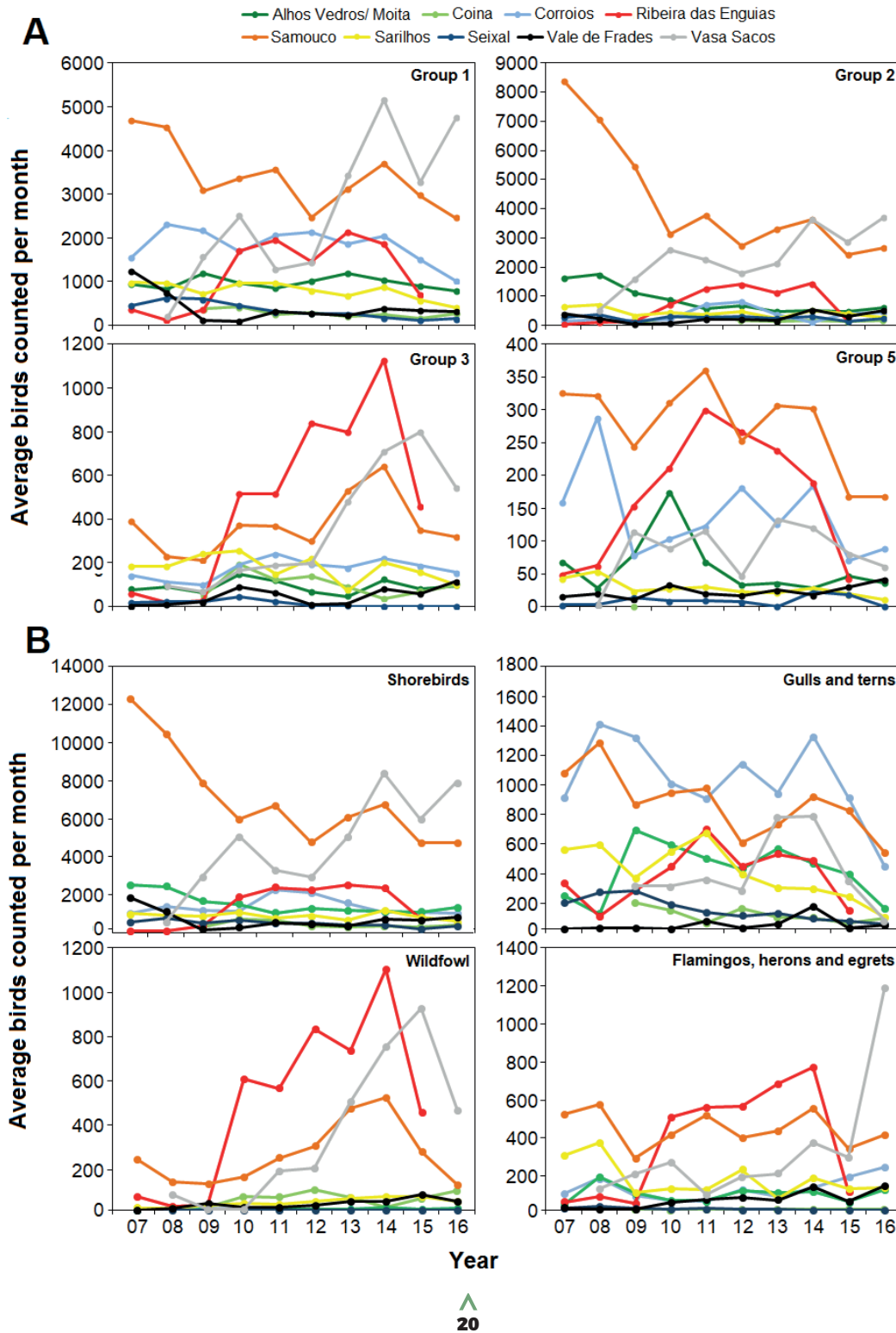


Figure 6 - Average waterbirds counted per month in each of the high tide roosts of the Tejo estuary between 2007 and 2016. We present data for the phenological groups defined in Figure 3 (Panel A) and for each of the main waterbird groups using the Tejo estuary's high tide roosts (Panel B).

Figura 6 - Média mensal de aves aquáticas contadas em cada refúgio de preia-mar do estuário do Tejo entre 2007 e 2016. São apresentados os dados referentes aos grupos fenológicos definidos na Figura 3 (Painel A) e para cada um dos principais grupos de aves aquáticas que usam os refúgios de preia-mar do Tejo (Painel B).



### Population trends from 2007 to 2016

We were able to derive trends for 22 wintering populations, seven of which had a stable population trend, eight were increasing and seven were decreasing (Table 4). For the remaining 18 wintering population we either obtained uncertain trends or the data was not sufficient to run the models (see Appendix 2 in the Supporting Information). For the three breeding populations analysed, black-winged stilts showed a stable trend, while little terns showed a moderate decrease and Kentish plover evidenced a strong decrease (Table 4). All analysed duck species showed strong increases, the same being true for great cormorant. Eurasian spoonbill *Platalea leucorodia* showed a moderate increase, while the two gulls, greater flamingo and little egret *Egretta garzetta* all show stable trends. The grey heron *Ardea cinerea* is declining at a moderate rate. Among shorebirds, trends varied considerable, ranging from strong decreases to strong increases (Table 4).

Count data for December and February mostly confirmed the trends obtained from January counts, although there are a few cases where they differed. For December, 12 species were in the exact same trend class, five changed to an adjacent trend class (either between a stable trend and a moderate trend, or between strong and moderate trends in the same direction) and in only two cases, black-tailed godwit and greater flamingo, we observed very different trends from those observed in January (more details in Appendix 2 of the Supporting Information). Additionally, for three species with a trend for January, data from December delivered an uncertain trend (see Appendix 2 in the Supporting Information). For February, 17 species showed the exact same trend class as in January, while five changed to an adjacent trend class (see Appendix 2 in the Supporting Information). In one case, the red knot, data from January delivered an uncertain trend, but data from February indicated a strong decrease ( $-0.273 \pm 0.011$ ;  $p < 0.001$ ; see Appendix 2 in the Supporting Information).

**Table 4** - Trends of waterbird populations in the Tejo estuary between 2007 and 2016. For each species we present the estimated annual change  $\pm$  SE, the P value indicating whether the estimated yearly change is significantly different from 0, the modelled change in numbers over the ten year period (decadal change), the population trend in the study areas (↓↓ strong decrease, ↓ moderate decrease, ⇒ stable, ↑ moderate increase, ↑↑ strong increase) and the most recent available flyway trend for the population using the Tejo estuary (Delany et al. 2009, Wetlands International 2017).

**Tabela 4** - Tendências populacionais das aves aquáticas do estuário do Tejo entre 2007 e 2016. Para cada espécie é apresentada a variação anual estimada  $\pm$  EP, o valor-P que indica se a variação anual é significativamente diferente de 0, a variação ao longo dos dez anos obtida a partir do modelo (decadal change), a tendência populacional nas áreas de estudo (↓↓ declínio acentuado, ↓ declínio moderado, ⇒ estável, ↑ crescimento moderado, ↑↑ crescimento acentuado) e a estimativa mais recente disponível para a tendência populacional ao nível da rota migratória (Delany et al. 2009, Wetlands International 2017).

SPECIES	ANNUAL CHANGE	P-VALUE	DECADAL CHANGE	TREND IN THE TEJO	FLYWAY TREND
<i>Calidris alpina</i>	-0.016 $\pm$ 0.001	<0.001	-13.5%	↓	Stable (ssp. alpina, 1997-2007)
<i>Limosa limosa</i>	-0.080 $\pm$ 0.003	<0.001	-52.8%	↓↓	Increasing (ssp. islandica, 1997-2007)
<i>Larus ridibundus</i>	0.010 $\pm$ 0.002	n.s.	+9.4%	⇒	Stable (W Europe, 1997-2007)
<i>Pluvialis squatarola</i>	0.059 $\pm$ 0.003	<0.05	+67.5%	↑↑	Decreasing? (ssp. squatarola from W Siberia and Canada, 1997-2007)

AIRO Waterbird monitoring in the Tejo estuary (2007-2016)

SPECIES	ANNUAL CHANGE	P-VALUE	DECADAL CHANGE	TREND IN THE TEJO	FLYWAY TREND
<i>Larus fuscus</i>	-0.007±0.004	n.s.	-6.1%	→	Increasing (ssp. <i>graellsii</i> from W Europe, 1990-2000)
<i>Phoenicopterus roseus</i>	-0.007±0.005	n.s.	-6.1%	→	Increasing (W Mediterranean, 1994-2004)
<i>Tringa totanus</i>	-0.008±0.003	<0.05	-7.0%	↓	Stable (ssp. <i>totanus</i> from N Europe, 1987-1997)
<i>Recurvirostra avosetta</i>	0.006±0.003	n.s.	+5.5%	→	Stable (W Europe, 1997-2007)
<i>Charadrius hiaticula</i>	0.076±0.012	<0.001	+93.3%	↑↑	Fluctuating (ssp. <i>hiaticula</i> , 1997-2007)
<i>Himantopus himantopus</i> (wintering)	-0.013±0.007	n.s.	-11.1%	→	Stable (SW Europe and NW Africa, 1990-2000)
<i>Himantopus himantopus</i> (breeding)	-0.005±0.005	n.s.	-4.4%	→	Stable (SW Europe and NW Africa, 1990-2000)
<i>Limosa lapponica</i>	0.044±0.010	<0.05	+47.3%	↑	Increasing (ssp. <i>lapponica</i> , 1997-2007)
<i>Anas crecca</i>	0.399±0.014	<0.001	+1953%	↑↑	Increasing (ssp. <i>crecca</i> form NW Europe, 1997-2007)
<i>Anas platyrhynchos</i>	0.149±0.013	<0.001	+249%	↑↑	Increasing (NW Europe and W Mediterranean, 1997-2007)
<i>Egretta garzetta</i>	-0.003±0.011	n.s.	-2.3%	→	Increasing (ssp. <i>garzetta</i> from W Europe, 1997-2007)
<i>Anas clypeata</i>	0.206±0.007	<0.001	+440%	↑↑	Increasing (NW Europe and W Siberia, 1997-2007)
<i>Numenius arquata</i>	-0.095±0.010	<0.05	-59.3%	↓↓	Decreasing (ssp. <i>arquata</i> , 1997-2007)
<i>Charadrius alexandrinus</i> (breeding)	-0.126±0.016	<0.05	-70.2%	↓↓	Unknown (ssp. <i>alexandrinus</i> from W Europe, 1996-2006)
<i>Arenaria interpres</i>	-0.089±0.009	<0.05	-56.8%	↓↓	Increasing? (ssp. <i>interpres</i> from NE Canada and Greenland, 1997-2007)
<i>Ardea cinerea</i>	-0.048±0.012	<0.05	-35.8%	↓	Increasing (ssp. <i>cinerea</i> from N and W Europe, 1997-2007)
<i>Phalacrocorax carbo</i>	0.110±0.021	<0.05	+156%	↑↑	Increasing (ssp. <i>carbo</i> from NW Europe, 1997-2007)
<i>Tringa nebularia</i>	-0.019±0.012	n.s.	-15.9%	→	Stable (N Europe, 1990-2000)
<i>Calidris minuta</i>	-0.234±0.016	<0.001	-90.9%	↓↓	Increasing? (N Europe 1997-2007)
<i>Platalea leucorodia</i>	0.093±0.021	<0.05	+123%	↑	Increasing (ssp. <i>leucorodia</i> from W Europe and Mediterranean, 1996-2006)
<i>Sternulla albifrons</i> (breeding)	-0.047±0.014	<0.05	-35.2%	↓	Decreasing (ssp. <i>albifrons</i> from W Europe and Mediterranean, 1990-2000)

### Proportion of shorebirds and wildfowl using remote saltmarshes

Remote saltmarshes were very important for wildfowl, with 75-100% of individuals of all analysed species using these areas as roosting sites (Table 5). Among shorebirds, for both pied avocet and bar-tailed godwit there were on average more birds in those saltmarshes than in all the monitored roosts combined

(Table 5), while for another six species numbers in remote saltmarshes represent over 30% of the average “total” number of birds using the estuary (Table 5). However, the proportion of birds using remote saltmarshes varied enormously, in cases such as bar-tailed godwit, red knot, curlew *Numenius arquata* and northern lapwing *Vanellus vanellus* ranging from 0% to nearly 100%).

**Table 5** - Shorebird and waterfowl species using remote saltmarshes of the Tejo estuary not regularly surveyed within the scope of this monitoring programme, based on boat counts performed by ICNF in the months of January, February and December from January 2012 to January 2015. For each species we present an estimate of the average “total” number of individuals present in the estuary in those months ( $\pm$  SD), by adding the birds counted in remote saltmarshes and those counted in the present study. The average percentage ( $\pm$  SD) of birds using the remote saltmarshes during those months and its range is also given. Species are ordered by decreasing average percentage using the remote saltmarshes.

Tabela 5 - Espécies de limícolas e de anatódeos que usam os sapais da zona nordeste do estuário do Tejo, que não foram monitorizados no presente estudo, com base nas contagens efectuadas de barco pelo ICNF nos meses de Janeiro, Fevereiro e Dezembro entre Janeiro de 2012 e Janeiro de 2015. Para cada espécie é apresentada uma estimativa média do “total” de indivíduos que usaram o estuário nesses meses ( $\pm$ DP), obtida somando as aves contadas pelo ICNF nesses sapais e as aves contadas durante a presente monitorização. É também apresentada a percentagem média ( $\pm$ DP) de aves que usaram os sapais não monitorizados, assim como a percentagem mínima e máxima durante os meses em questão. As espécies estão ordenadas por ordem decrescente da percentagem média que usa os sapais não monitorizados.

SPECIES	ESTUARY “TOTAL”	PERCENTAGE IN REMOTE SALTMARSHES
<i>Anas acuta</i>	7117 $\pm$ 9047	100 $\pm$ 0 (100-100)
<i>Netta rufina</i>	7 $\pm$ 8	100 $\pm$ 0 (100-100)
<i>Anser anser</i>	2528 $\pm$ 2016	98.9 $\pm$ 2.9 (91-100)
<i>Anas penelope</i>	4471 $\pm$ 1996	94.8 $\pm$ 7.4 (77-100)
<i>Tadorna tadorna</i>	285 $\pm$ 186	94.3 $\pm$ 8.7 (78-100)
<i>Anas platyrhynchos</i>	7684 $\pm$ 5199	87.6 $\pm$ 13.1 (65-99)
<i>Anas strepera</i>	108 $\pm$ 50	86.5 $\pm$ 13.5 (67-100)
<i>Aythya fuligula</i>	13 $\pm$ 14	86.2 $\pm$ 30.9 (31-100)
<i>Anas clypeata</i>	8581 $\pm$ 6335	79.9 $\pm$ 25.5 (32-97)
<i>Anas crecca</i>	7023 $\pm$ 3132	76.0 $\pm$ 12.1 (59-98)



SPECIES	ESTUARY "TOTAL"	PERCENTAGE IN REMOTE SALTMARSHES
<i>Recurvirostra avosetta</i>	5370±1984	64.4±16.1 (31-83)
<i>Limosa lapponica</i>	949±923	53.2±35.1 (0-100)
<i>Numenius phaeopus</i>	98±82	48.9±37.5 (0-93)
<i>Calidris alpina</i>	12595±2762	42.4±9.2 (26-55)
<i>Calidris canutus</i>	662±691	34.9±42.9 (0-98)
<i>Numenius arquata</i>	565±153	34.1±24.0 (2-69)
<i>Vanellus vanellus</i>	80±102	31.1±46.8 (0-100)
<i>Pluvialis squatarola</i>	4340±1992	30.3±18.8 (11-61)
<i>Calidris alba</i>	127±129	28.4±32.5 (0-87)
<i>Tringa totanus</i>	1721±571	26.1±13.6 (14-55)
<i>Limosa limosa</i>	3499±2104	20.6±31.0 (0-87)
<i>Arenaria interpres</i>	244±102	17.4±12.9 (0-36)
<i>Charadrius hiaticula</i>	793±503	16.0±24.4 (0-59)
<i>Tringa nebularia</i>	105±20	10.7±16.1 (0-42)
<i>Actitis hypoleucos</i>	34±11	7.1±4.7 (0-14)
<i>Charadrius alexandrinus</i>	236±271	7.0±15.0 (0-43)
<i>Himantopus himantopus</i>	476±80	0.2-0.4 (0-1)
<i>Calidris minuta</i>	70±58	0.0-0.0 (0-0)
<i>Calidris ferruginea</i>	57±63	0.0-0.0 (0-0)
<i>Tringa erythropus</i>	50±29	0.0-0.0 (0-0)
<i>Gallinago gallinago</i>	19±12	0.0-0.0 (0-0)
<i>Haematopus ostralegus</i>	12±14	0.0-0.0 (0-0)
<i>Tringa ochropus</i>	8±5	0.0-0.0 (0-0)
<i>Philomachus pugnax</i>	4±7	0.0-0.0 (0-0)

## Discussion

### Phenological patterns of the waterbird community in the Tejo estuary

The present study confirms the international importance of the Tejo estuary for waterbirds both as a wintering area and as a refuelling site during migratory periods (e.g. Alves et al. 2011, Catry et al. 2011, Rocha et al. 2017). In fact, most species show clear peaks associated with migratory periods, particularly in autumn as previously described (Alves et al. 2011, Catry et al. 2011).

The five phenological groups defined through the cluster analysis also corroborate these trends, with the largest group being associated with species for which the counts from months in late summer and autumn rank among the highest. Smaller groups of species were associated with higher counts in mid-winter and during spring. Species that breed in the estuary or use it during post-breeding movements were also grouped separately, but Kentish plover, which breeds in some of the studied high tide roosts (Rocha et al. 2016), was included among the group of species with a clear autumnal migratory peak. This species peaks in July-September and in October-January numbers remain higher than those observed during the breeding season (see Appendix 1 in the Supporting Information). This pattern is similar to that observed in other Portuguese wetlands (e.g. Batty 1992, Lopes et al. 2005) and suggests that most Kentish plovers using the estuary do not belong to the local breeding population. Still, we cannot rule out that the more cryptic behaviour exhibited by Kentish plovers during the breeding season may influence our ability to accurately estimate their numbers when nesting.

Overall, it is quite clear that besides being a critical wintering area for many waterbirds, many species strongly depend on the Tejo estuary during migratory passage, highlighting the key role of this wetland for the migratory fluxes linking northern

Europe and Africa in spring and, especially, in autumn. Although there is considerable variation among different species, at least for shorebirds this trend for higher counts during autumn migration is also observed in several other wetlands along the flyway (e.g. Batty 1992, Le Drean Quenec'hdu et al. 1995, Scheiffarth & Becker 2008, El Hamoumi & Dakki, 2010). Despite some exceptions (e.g. Lopes et al. 2005, Lourenço 2006), this pattern is most likely explained by the fact that in spring migrants are expected to be more time-stressed in order to arrive early at their breeding areas, thus making fewer and shorter stops along the way (Lindström & Alerstam 1992, Farmer & Wiens 1998), meaning that monthly counts performed in spring are more likely to miss very short migratory peaks that can last just a few days. In fact, counts performed at shorter intervals in the Tejo evidenced the occurrence of such short migratory peaks during spring (Catry et al. 2017, Catry et al. unpub. data).

### Relative importance of different roosts

Despite some variation over the years, it is clear that a small number of roosts support the vast majority of waterbirds using the Tejo estuary throughout the annual cycle. Together, Samouco, Vasa Sacos, Ribeira das Enguias, Corroios and Alhos Vedros/Moita support over 80% of all counted individuals, and this is true across all waterbird groups. The first three roosts are located within the local protected areas, but only Samouco is partially managed for waterbird conservation. As for the other two key roosts (Corroios and Alhos Vedros/Moita), they lie outside the protected areas and therefore have no legal protection. The fact that roughly 30% of waterbirds in the Tejo estuary use roosts with no legal protection (Table 3) raises concern, as these areas are likely more vulnerable to urban and industrial development. In fact, for species such as curlew (82%),

whimbrel *Numenius phaeopus* (66%), lesser black-backed gull (62%), pied avocet (57%), black-headed gull (53%) and common redshank (50%) over half the population is found in roosts with no legal protection. Furthermore, although there is evidence that waterbirds can switch roosts when facing local changes (e.g. Burton et al. 1996, Rocha 2015), all of these unprotected sites are clustered around the southern part of the estuary, where there are no nearby alternative roosts. Since shorebirds prefer to forage near roosting sites (e.g. Dias et al. 2006), any anthropogenic pressure to these unprotected roosts could jeopardize the availability of the southern part of the estuary as a suitable foraging area for shorebirds.

Overall we can divide the studied roosts in two main categories, saltpans and saltmarshes, and the current trends point for a decline in the use of saltpans and an increase in the use of saltmarshes. Even in Vasa Sacos, which showed an increase in bird numbers in the last years, most birds now use the adjacent saltmarsh and not the saltpans. This mainly results from the fact that saltpans are no longer used for salt production, becoming either abandoned and increasingly encroached by vegetation, or flooded when converted to shrimp farms, both of which are less likely to provide favourable roosting condition for shorebirds.

In fact, saltmarshes were likely the main roosting option for shorebird before man-made saltpans became available. With few exceptions, most shorebird species also use the remote saltmarshes in the north-eastern area of the estuary as roosting sites during winter. In fact, these few exceptions can in most cases be attributed to the difficulties of identifying birds while doing counts from a boat. Good examples are little stint *Calidris minuta* and curlew sandpiper *C. ferruginea* which can easily be missed within large flocks of dunlin. As previously mentioned, these boat count data are difficult to interpret, since they are very rough esti-

mates based on information that may not be directly comparable with our counts, and the proportion of birds that use these remote saltmarshes varies greatly over time. In any case, based on the information gathered by ICNF counts, it is clear that a significant proportion of shorebirds, and a much larger proportion of wildfowl were not fully covered during our land counts. However, if we assume that the use of remote saltmarshes, despite its apparent fluctuations, remains mostly similar over time, this issue should not affect our main findings regarding shorebird phenological patterns, relative use of different (monitored) roosts and population trends over the studied decade.

### Waterbird population trend in the Tejo estuary

Available data suggests that different waterbird species and groups are facing distinct population trends in the East Atlantic Flyway. Whereas wildfowl, gulls and terns mostly show favourable trends, a large proportion of shorebird populations are currently declining (Davidson & Stroud 2006, Stroud et al. 2006, van Roomen et al. 2015). Previous data from the Tejo estuary, referring to the period between 1975 and 2006, indicated that three of the five most abundant shorebirds: dunlin, grey plover and redshank, were declining locally (Catry et al. 2011). At least to some extent, these declines are believed to be driven by local factors as the same trends were neither observed in other Portuguese wetlands nor at the flyway scale (Catry et al. 2011).

Our present data, referring to the last ten years, confirm the local declining trend for dunlin and redshank, while grey plover now exhibited a strong increase. These data also confirm a stable trend for pied avocet, but black-tailed godwit, which until 2006 had a stable trend (Catry et al. 2011), is now in strong decline. This change in the local trend for black-tailed godwit may in part result

from an increasing proportion of the estuarine population, which is mostly composed of Icelandic godwits (*L. l. islandica*), using nearby rice fields more regularly (Alves et al. 2010). These rice fields represent the main feeding grounds for Continental godwits (*L. l. limosa*), which winter in West Africa and start migrating through the Iberian Peninsula in January (Alves et al. 2010, Lourenço et al. 2010), but in recent years observation of colour-marked birds evidenced that the number of Icelandic godwits using this alternative habitat has increased (J.A. Alves unpub. data). Furthermore, some of these rice fields are kept flooded during winter to benefit birds, so they may also receive higher number of other species, such as dabbling ducks and flamingos, than in the past.

Globally, the present shorebird trends in the Tejo estuary suggest that high-Arctic breeders are doing better than species that breed at lower latitudes. Among the species from the high-Arctic, ringed plover and grey plover both showed strong increases and bartailed godwit showed a moderate increase, but ruddy turnstone *Arenaria interpres* and little stint both showed strong decreases. Among the species that breed at lower latitudes, black-tailed godwit and curlew showed strong decreases; dunlin and common redshank showed moderate decreases, while only pied avocet, black-winged stilt and common greenshank evidenced stable trends.

For most of the analysed populations, the trend we observed in the monitored high tide roosts fits with the available data on flyway trends (Table 4). However, there are nine cases in which local trends differ from flyway trends. In part, these different trends can result from a temporal mismatch between different data sources, as the most recent available flyway trends at best refer to the decade immediately preceding the collection of our data and may thus not be directly comparable. Still, other factors may be at play here. Lesser black-backed gull, greater flamingo and little egret evidenced

a stable trend in the Tejo, whereas at the flyway scale these populations are believed to be increasing. Grey plover and common ringed plover, which increased strongly in the Tejo, appear to be fluctuating or even decreasing at the flyway scale. Dunlin and common redshank, which showed a moderate decrease in the Tejo, are considered stable at the flyway scale, whereas black-tailed godwit and ruddy turnstone, which showed strong decreases within the Tejo estuary, and grey heron with a moderate decrease, are all believed to be increasing at the flyway scale (Table 3). These mismatches between local and global trends are also observed in other wintering and staging sites across Europe (e.g. Austin et al. 2000, Eybert et al. 2003, Lopes et al. 2005, Meltofte et al. 2006) and may result from local and regional processes (like the black-tailed godwit example mentioned above), from local changes in food availability (e.g. Austin et al. 2000), human disturbance (Eybert et al. 2003), or from large-scale changes in the overall distribution of wintering and migratory populations along the flyway (e.g. Austin & Rehfish 2005, Rakhimberdiev et al. 2010). Although unlikely, we can also not rule out the possibility that changes in the proportion of birds using the saltmarshes that were not monitored in this study may influence the observed population trends. The cases of dunlin, redshank and ruddy turnstone may be of particular concern as the decreases in the Tejo go against the global trend and may indicate these species are facing problems locally, as had been previously described for the first two species since the 70s (Catty et al. 2011).

Such difficulties in interpreting and reconciling local and global trends evidently reinforces the need for having reliable and up-to-date information on the status of waterbird populations across the flyway, which can only be obtained through long-term monitoring programmes such as the one described here for the Tejo estuary.

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# Feral cat control and the population recovery of the endemic St Helena Plover

## Controlo de gatos assilvestrados e recuperação populacional de uma espécie endémica, o Borrelho de Santa Helena

Dennis Leo<sup>1</sup>, Kevin George<sup>1</sup>, Gavin Ellick<sup>1</sup>, Steffen Oppel<sup>2\*</sup>

<sup>1</sup> St Helena National Trust, Broadway House, Jamestown, St Helena, South Atlantic

<sup>2</sup> RSPB Centre for Conservation Science, Royal Society for the Protection of Birds, The David Attenborough Building, Cambridge CB2 3QZ, United Kingdom

\* **Corresponding author:** [steffen.oppel@rspb.org.uk](mailto:steffen.oppel@rspb.org.uk)



## ABSTRACT

Invasive non-native species are one of the greatest drivers of biodiversity loss on islands worldwide. Eradication of invasive species is an efficient way to restore island ecosystems, but on inhabited islands it can be problematic to eradicate species that are valued by humans, such as cats (*Felis catus*). Permanent control of only the feral segment of such species may nonetheless have beneficial effects on native biodiversity. Here we describe the population recovery of the St Helena Plover (*Charadrius sanctaehelenae*) after feral cat control was introduced on the island in 2012. We counted the global population every year and monitored the success of 655 nests between 2011 and 2018. Since 2012 a total of 183 feral cats were removed in 12,559 trap nights in the two key breeding areas of the St Helena Plover. The breeding success in those two areas increased from 6.4% to 30.1% of pairs being able to raise at least one fledgling. The St Helena Plover population increased from 364 in 2011 to 627 adult individuals in 2018. Although cat trapping effort has been significantly reduced since 2012, the plovers' mean breeding success remained >15% in all years since 2012. Simultaneously implemented activities to reduce the number of fertile house cats and an awareness raising campaign to reduce the number of released cats have likely also contributed to a lower predation pressure. Overall, this example demonstrates that a globally threatened bird species can be safeguarded by targeted feral cat control.

**Keywords:** invasive species, island restoration, predator control, shorebird, wader.

## RESUMO

As espécies não-nativas invasoras são uma das maiores causas de perda de biodiversidade em ilhas por todo o mundo. A erradicação de espécies invasoras é uma maneira eficiente de restaurar ecossistemas insulares, mas em ilhas habitadas pode ser problemático erradicar espécies que são valorizadas pelo Homem, como o gato (*Felis catus*). O controlo permanente de animais assilvestrados pode, no entanto, ter efeitos benéficos sobre a biodiversidade nativa. Neste estudo, descrevemos a recuperação populacional do Borrelho de Santa Helena (*Charadrius sanctaehelena*) após o início do controlo de gatos assilvestrados na ilha de Santa Helena, em 2012. Foram realizadas contagens anuais da população global de borrelhos e monitorizado o sucesso de 655 ninhos entre 2011 e 2018. Desde 2012, um total de 183 gatos assilvestrados foram removidos ao longo de 12,559 noites de armadilhagem nas duas principais áreas de reprodução do borrelho de Santa Helena. O sucesso reprodutor dos borrelhos nessas duas áreas aumentou de 6.4% para 30.1% de casais capazes de criar pelo menos uma cria. Como consequência, a população do Borrelho de Santa Helena aumentou de 364 em 2011 para 627 em 2018. Embora o esforço de captura de gatos tenha sido reduzido significativamente desde 2012, o sucesso de reprodução permaneceu elevado. Atividades simultaneamente implementadas para reduzir o número de gatos domésticos férteis e uma campanha pública de consciencialização para reduzir o número de gatos libertados terão também provavelmente contribuído para uma menor pressão de predação. Este exemplo demonstra que uma espécie de ave globalmente ameaçada pode ser protegida pelo controlo de gatos assilvestrados.

**Palavras-chave:** ave limícola, controlo de predadores, espécies invasoras, restauro insular.

## Introduction

Invasive non-native species are one of the greatest drivers of human-caused loss of biodiversity worldwide (Simberloff et al. 2013, Maxwell et al. 2016), and many species that are endemic to islands are threatened by invasive species (Spatz et al. 2017). Mammalian predators like cats *Felis catus* have been introduced to many islands, and have caused the decline or extinction of many island-endemic species (Medina et al. 2011, Nogales et al. 2013). Over the past 40 years, conservation managers have eradicated invasive mammals from many islands to benefit globally threatened biodiversity (Jones et al. 2016). However, especially on islands with human populations that value some invasive species as domestic companions or as live-

stock, eradication of invasive species can be complicated (Ogden & Gilbert 2009, Opper et al. 2011, Glen et al. 2013). Long-term control of invasive cats, where the population is reduced but not eliminated, may nonetheless be a practically feasible and potentially beneficial conservation management option on such islands (Smith et al. 2010, Hartway & Mills 2012, Cruz et al. 2013).

The St Helena Plover *Charadrius sanctaehelena* is a globally threatened wader that inhabits natural semi-desert and anthropogenic pasture habitats on the island of St Helena, a United Kingdom Overseas Territory in the South Atlantic Ocean. Since its discovery by Portuguese sailors in 1502, St Helena has lost six endemic bird species likely due to

the effects of introduced mammals (Ashmole 1963), and the St Helena Plover declined substantially between the 1980s and 2000s as a consequence of agricultural changes and the effects of feral cat predation (McCulloch 1991, McCulloch 2009, Burns 2011, Burns et al. 2013a). After initial concerns over habitat loss caused by the abandonment of pastures and subsequent vegetation succession (McCulloch & Norris 2001, McCulloch 2009), more intensive grazing was initiated to maintain or restore the habitat quality of some pasture habitats. However, predation by introduced feral cats was sufficiently high to potentially explain population declines (Burns 2011, Burns et al. 2013a), which triggered additional research to reduce predation pressure (Burns et al. 2013b). From 2011-2013 an experimental study was conducted to examine whether feral cat trapping could benefit the St Helena Plover, and this study concluded that breeding success would increase especially in semi-desert habitats (Oppel et al. 2014). However, this experiment could not evaluate whether the increase in breeding success would be sufficient to lead to a population recovery.

Here we describe the population recovery of the St Helena Plover that started in 2006 and ultimately led to the species being down-listed from ‘Critically Endangered’ to ‘Vulnerable’ in 2016. We monitored the global population by annual counts across the entire island, measured breeding success in two key breeding areas between 2011 and 2017, and document the cat trapping effort that was maintained following the initial cat control experiment. Because other conservation measures, such as intensive grazing to maintain pasture habitat quality and public education and ordinances to reduce the release of domestic cats into the wild, occurred simultaneously to feral cat control since 2012, our study does not constitute a controlled scientific experiment that can demonstrate the causal effect of feral cat control.

## Methods

### Study area

St Helena is a volcanic island located in the South Atlantic (15°58’S, 5°43’W) with an area of 124 km<sup>2</sup> and a human population of approximately 4,500 people. The endemic St Helena Plover inhabits various treeless habitats, mainly grazed farmland and open sandy plains of a semi-desert, and the global population of this species is distributed among 32 habitat patches. The original cat removal experiment focussed on 250 ha of sheep grazed grasslands at the south-western tip of St Helena, and 215 ha of semi-desert in the eastern part of the island, which together hosted around 30% of the global population (Burns 2011, Oppel et al. 2014). Since 2012, the semi-desert area has been extensively modified due to the construction of an airport, leading to large-scale re-distribution of rock and soil, levelling of a broad and formerly undulating rocky area, and the construction of roads, terminal facilities, and a runway. Although this construction led to major changes in the habitat, a mitigation protocol was in place that mandated that the construction company restored suitable habitat elsewhere on the island, avoided loss of any St Helena Plovers through nest monitoring and local deferral of disturbing activities, and the implementation of feral cat control.

Here we focus on two sites where cat trapping and monitoring were continued until 2017, the semi-desert area included in the original study (“Prosperous Bay”) where airport construction occurred, and a pasture area (“Deadwood”) c. 3 km to the north, which encompasses c. 320 ha of pasture and holds the second largest congregation of St Helena Plovers (McCulloch & Norris 2001, McCulloch 2009), but was not included in the experimental study of cat control (Oppel et al. 2014).

### Feral cat trapping and removal

The density of feral cats on St Helena was estimated at 3.9 cats/km<sup>2</sup> in our study area in the semi-desert, and 8.7 cats/km<sup>2</sup> at another study area with similar pasture habitat as Deadwood, during an intensive study in 2011 before cat control began (Oppel et al. 2014). In addition, St Helena inhabitants own over 1,000 domestic cats, most of them free-ranging. Since 2012, a public ordinance requires pet owners to have their cat registered and micro-chipped, which allowed instant identification of domestic cats in the field. We used live cage trapping so that any domestic cats could be released, which was labour-intensive but critical to maintain public support (Ratcliffe et al. 2010, Oppel et al. 2014). Cat trapping was initiated in May 2012, and traps were baited with food, placed where cats had been previously seen or caught, and checked daily. Trapping effort and trap placement varied among years depending on available staff, but traps were generally placed in areas where capture

probability was considered to be highest based on past experience. Trapped cats were first checked for micro-chips and compared against a photographic library of domestic cats to avoid euthanasing domestic cats. Non-registered feral cats were euthanased unless their behaviour allowed domestication. The control procedures were carried out by staff trained by a veterinary surgeon and approved by the state veterinary service of St Helena.

Intensive feral cat trapping continued until September 2012, after which trapping was considerably reduced. Due to the daily trap checks, intensive feral cat control could not be sustained constantly, hence trapping focussed on the peak season of the breeding period between September and March at each site, and details on the cat trapping effort are provided in Table 1. Feral cat control also occurred intermittently at other sites on St Helena between 2012 and 2017, but these efforts are not described here because they do not correspond to available monitoring data of St Helena Plovers.

**Table 1** - Cat trapping effort in two key breeding areas of St Helena Plover on the island of St Helena between 2012 and 2017. For each breeding ‘season’ (generally between September and March of the following year) we provide the trapping effort and the number of cats captured (both feral and domestic), plus the percentage of captured cats that were domestic pets (identified by photos or implanted transponder, a legal requirement since 2012).

**Tabela 1** - Esforço de captura de gatos nas duas principais áreas de reprodução do Borrelho de Santa Helena na ilha de Santa Helena entre 2012 e 2017. Para cada época de reprodução (que decorre maioritariamente entre Setembro e Março do ano seguinte) são apresentados o esforço de armadilhagem, o número total de gatos capturados e a percentagem de gatos domésticos capturados (identificados através de fotografias ou da presença de micro-ship).

SITE	SEASON	MONTHS OF TRAPPING EFFORT	TOTAL TRAPPING EFFORT (TRAP NIGHTS)	TOTAL NUMBER OF CATS CAPTURED	% DOMESTIC PETS	CATS CAPTURED / 100 TRAP NIGHTS
<i>Deadwood</i>	2012-2013	16	2586	43	16.3	1.66
	2013-2014	12	1302	18	11.1	1.38
	2014-2015	0	0	0	---	0
	2015-2016	2	120	2	0	1.67
	2016-2017	7	276	7	14.3	2.54

SITE	SEASON	MONTHS OF TRAPPING EFFORT	TOTAL TRAPPING EFFORT (TRAP NIGHTS)	TOTAL NUMBER OF CATS CAPTURED	% DOMESTIC PETS	CATS CAPTURED / 100 TRAP NIGHTS
<i>Prosperous Bay</i>	2012-2013	16	2119	36	16.7	1.70
	2013-2014	12	1648	36	16.7	2.18
	2014-2015	10	1397	19	0	1.36
	2015-2016	12	2042	28	0	1.37
	2016-2017	12	1069	16	0	1.50

### Nest monitoring and Calculation of Breeding success

St Helena Plovers can nest year-round, but the majority of the population breeds at least once between September and March without a pronounced peak with synchronised breeding activity (McCulloch 1991, Burns 2011). Nest monitoring followed established procedures for this species (Burns et al. 2013a, Oppel et al. 2014), which consisted of continuously searching for nests by following behavioural cues of adult birds between September and March of the following year. Once found, nests were visited approximately weekly and were checked from a distance to avoid disturbance during incubation, which typically lasts 28 days (McCulloch 1991, Burns 2011). Once chicks had hatched, the area where adults would lead their dependent young was checked at least once per week until chicks had grown wing feathers and were able to fly, usually at the age of 36 days (McCulloch 1991, Burns 2011). Nests were classified as successful if at least one chick survived until fledging, and classified as unsuccessful if the nest scrape was found empty during incubation or if parent birds were encountered without dependent young during the time of chick rearing.

Due to the variable monitoring effort we were not able to estimate nest survival, and therefore calculated breeding success as the proportion of nests that managed to raise at least one fledgling. We then estimated the effect of feral cat control (as a binary variable) using a generalised linear mixed model with breeding success as a binary response variable, site and cat control as fixed factors, and year as a random effect to account for annual variation in weather and other events that may affect breeding success. We fit this model in R3.4.2 using the package lme4 (Bates et al. 2014), and we present mean and 95% confidence intervals of model-derived breeding success estimates for all years and both study sites.

### Monitoring of global population size

Censuses of the global population of St Helena Plovers followed established protocols (McCulloch & Norris 2001, McCulloch 2009) and occurred in January each year, when 32 census areas were visited once during a three-week period. St Helena Plovers are territorial and easily detectable in their open breeding habitat, and January is the time of year with the greatest proportion of the pop-

ulation nesting and therefore tightly associated with their territories (McCulloch 1991, McCulloch 2009). Observers walked along transects that ensured that all areas within a census block were approached to within 25 m to ensure that no birds were missed during the surveys. Observer teams (2-3 people), identity of observers, and the observation effort were similar in all years, hence we did not expect systematic variation in count data due to observer effects. Because St Helena Plovers are reluctant to fly, and generally only fly very short distances when accidentally disturbed, the risk of double-counting the same birds was relatively small (McCulloch 2009). We classified encountered birds into three age categories, namely breeding adult, recently fledged juvenile and chicks prior to fledging. We present the total number of adults only since recently fledged juveniles may be more mobile and may move between census areas.

We estimated the population trend for the global population by using a hierarchical state-space model for population counts, which decomposed the monitoring data into a population process and an observation error component (de Valpine 2003, Clark & Bjørnstad 2004, Kéry & Schaub 2012). State-space models allow a more accurate estimate of population trend than standard linear models because they explicitly allow for environmental stochasticity (Wilson et al. 2011, Kéry & Schaub 2012). Our model fitted the log-transformed count data to a normal distribution, and allowed for normally distributed observation errors around each annual count. We implemented the state-space model in a Bayesian framework to properly account for error propagation when summing population sizes and averaging growth rates over time. We fitted the state-space models using Markov Chain Monte Carlo methods in NIMBLE v. 0.6-11 (de Valpine et al. 2016). We ran four Markov chains each with 50,000 iterations and discarded the first 35,000 iterations to ensure that the

models converged. Convergence was tested using the Gelman-Rubin diagnostic (Brooks & Gelman 1998), and all parameters had values of  $R\text{-hat} < 1.02$ . We report estimated global population size for each year and the mean annual population growth rate as the posterior median and 95% credible intervals (Kéry & Schaub 2012).

## Results

Between May 2012 and August 2017 we captured 183 feral cats and 22 domestic pet cats during 12,559 trapping nights in the two focal breeding areas (Table 1). On average, we captured 1-2 feral cats per 100 trap nights, and this capture rate did not appear to decline markedly over the years (Figure 1), presumably because cats continuously re-invade our controlled sites from surrounding populations.

Despite the gradual decline in trapping effort and the number of cats removed every year (Table 1), the breeding success of St Helena Plovers remained relatively high since the feral cat trapping was initiated in May 2012 (Figure 1). We monitored a total of 655 nests ( $n = 390$  at Deadwood, and  $n = 265$  at Prosperous Bay), and the average breeding success prior to feral cat trapping was 6.5%, but increased to 30.5% since 2012 (Figure 1). The model accounting for site and random year effects found a positive effect of feral cat control on breeding success ( $\beta = 1.91$ , standard error = 0.61,  $p = 0.002$ ; Figure 1).

The global population of St Helena Plovers decreased between 1999 and 2006 (McCulloch 2009), but increased over the past 12 years to population sizes that exceeded any previous records (Figure 2). The long-term population growth rate was estimated to be 2.1% (95% credible interval -7.1 – 11.2%), and the largest annual growth rate was estimated between 2015 and 2016 when the population increased by 19.1% (-3.1 – 37.1%,

**Figure 1** - Estimated breeding success of St Helena Plovers and number of feral cats captured per 100 trap nights in two study areas on St Helena between 2011 and 2017. Breeding success is shown with 95% confidence intervals derived from a generalised linear mixed model accounting for site and year effects; numbers above the x-axis indicate the number of monitored nests per site and year.

**Figura 1** - Estimativas do sucesso reprodutor do borrelho de Santa Helena e número de gatos assilvestrados capturados por 100 armadilhas.noite nas duas áreas de estudo da ilha de Santa Helena entre 2011 e 2017. O sucesso reprodutor é apresentado com intervalos de confiança de 95% estimados a partir de um modelo linear generalizado misto em que o local e o ano foram incluídos como factores. Os valores sobre o eixo do x indicam o número de casais monitorizados em cada local e cada ano.

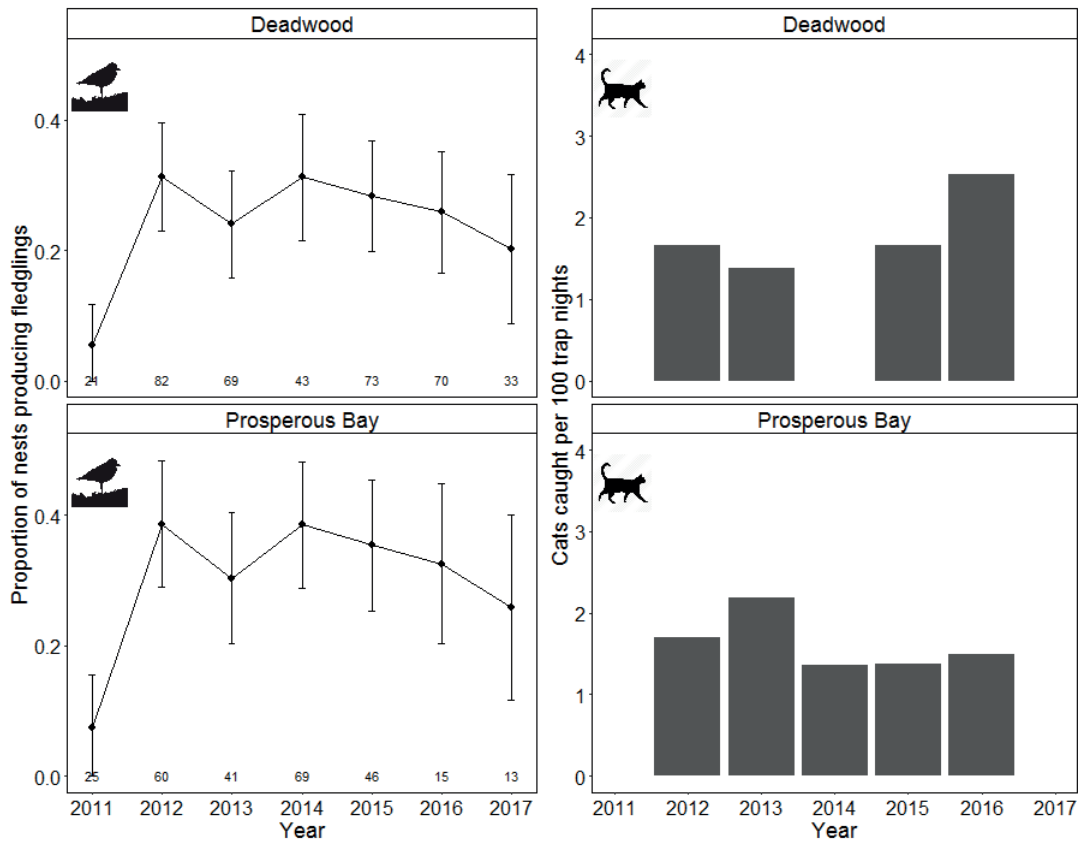


Figure 2). The global population exceeded 600 adult birds for the first time in 2018 with a raw count of 627 birds and a population estimate of 619 (486 – 729) birds (Figure 2).

## Discussion

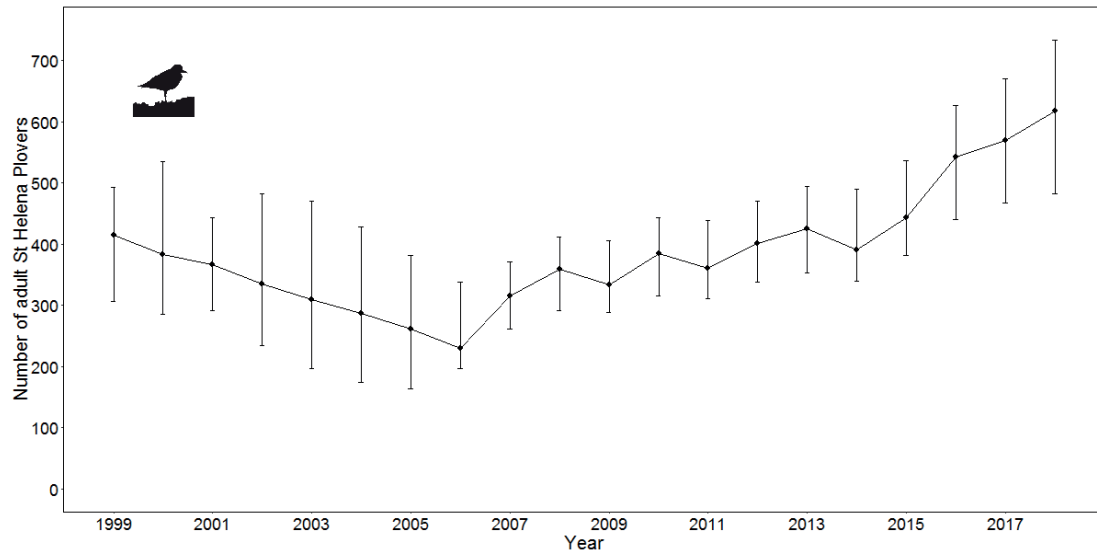
The St Helena Plover population has recovered since its historic low point in 2006 as a consequence of substantial conservation efforts. These conservation efforts included habitat restoration and the resumption of

intensive grazing regimes that led to shorter vegetation and thus more suitable habitat (McCulloch & Norris 2001, McCulloch 2009). In addition, feral cat control and other measures such as public awareness campaigns aiming to increase the number of neutered domestic cats and the reduction of the common practice to release unwanted kittens into the wild were implemented on St Helena to limit the size of the feral cat population (Burns et al. 2013a, Oppel et al. 2014). We have shown here that the breeding success of St Helena Plovers increased roughly 5-fold since



**Figure 2** - Estimated global population size of adult St Helena Plovers between 1999 and 2018. Population size was estimated with a state-space model accounting for environmental stochasticity and random observation error; error bars represent 95% credible intervals.

**Figura 2** - Estimativa da população de Borrelho de Santa Helena (número de aves adultas) entre 1999 e 2018. O número de indivíduos foi estimado com base em modelos de estados de espaço que consideram a estocasticidade ambiental e erros aleatórios de observação. As barras de erro representam intervalos de credibilidade de 95%.



feral cat control was introduced, and we therefore believe that this sustained effort to suppress feral cat populations has contributed to the recent population recovery of the species.

Feral cat control is unlikely to be the sole reason for the population recovery. The recovery started several years before feral cat control was initiated, and this early recovery was most likely due to the restoration of abandoned pastures. In addition, legislation in effect since 2012 mandates that all domestic cats must be registered and micro-chipped, and there is a financial incentive for cat owners to have their pets neutered, which may have also led to a lower feral cat population on the island in general, although this effect cannot be quantified. However, the period of feral cat control coincided with the construction of an airport in one of the core breeding habitats of the St Helena Plover, which could have potentially had a very negative effect on the population. Despite the destruction of several

hundred hectares of formerly suitable habitat and the disturbance of traffic and explosives, the plover population continued its upward trajectory throughout the construction of the airport. While feral cat control and construction activities occurred simultaneously and therefore have confounding effects on plover population size, we believe that it is possible that feral cat control effectively mitigated against the negative effects of habitat destruction caused by the construction of the airport.

Predator control operations are often logistically challenging and expensive (Moseby et al. 2009, Bengsen et al. 2012), which limited the intensity of the feral cat control in our project. Maintaining, setting and checking traps and humanely euthanasing cats, as well as liaising with cat owners to raise awareness and minimise risks to domestic cats would have required two full time staff members, but since 2014 only one staff member was available for this task. Hence, cat control efforts

were reduced (Table 1) and focussed on the peak breeding period between September and March. Despite the reduction in overall effort, the breeding success of St Helena Plovers at the two study sites remained higher in all years since 2012 than in 2011 (Figure 1) and in previous years before feral cat control was initiated (McCulloch 1991, Burns 2011, Burns et al. 2013a). Ongoing re-colonisation of study areas by feral cats is a common concern in any control operation and may reduce the beneficial effects of feral cat control (Bengsen et al. 2011, Bengsen et al. 2012). However, St Helena Plover nesting success did not decrease despite reduction in feral cat trapping effort, and we offer two plausible explanations for this counterintuitive pattern.

Our data suggest that cats were captured at similar rates (1-2 cats/100 trap nights) over time regardless of trapping effort (Table 1), which could suggest regular re-colonisation and relatively constant feral cat densities over time despite the control efforts. However, this similar trapping efficiency is confounded by efforts to deploy traps in areas most likely to maximise capture probability, hence it is in theory possible that capture rates were maintained despite a lower feral cat density. Unfortunately, feral cat density is very difficult and labour-intensive to estimate (Kéry et al. 2011, Oppel et al. 2012, Oppel et al. 2014), hence no independent assessment of cat population size over time is available for our study areas.

Another potential explanation for the elevated breeding success despite reduced trapping effort is the removal of specialised individual cats. Predation of threatened species such as the St Helena Plover may not be a trait common to all individual feral cats on St Helena, hence the cat control experiment starting in 2012 may have initially removed individual cats that had specialised on predating plover nests or chicks (Hervías et al. 2014, Moseby et al. 2015). If feral cats that subsequently colonised our study areas had different dietary or hunting preferences, then breeding success

of plovers may have remained relatively high regardless of cat density returning gradually to previous levels as long as none of the re-colonising cats were specialist predators of St Helena Plovers (Moseby et al. 2015). Because plover eggs or chicks are unlikely to form a major dietary component even for specialised predators, identifying the specialists and targeting them for capture and removal is unlikely to be practically feasible. Hence, while predator profiling and selective removal may in theory result in similar benefits to native biodiversity as blanket feral cat control (Moseby et al. 2015), this approach is unlikely to be practical for the conservation of St Helena Plovers.

We therefore encourage both the UK and the St Helena Governments to support continuing measures to reduce the pressure of domestic cats on wildlife (Calver et al. 2011, Loss et al. 2013). The long-term implementation of cat control will require consistent financial input and will only be an economic solution once the source of feral cats from unwanted domestic cats can be curtailed. Although there is much public controversy over the need for the control of feral cat populations (Loss & Marra 2018), there is overwhelming scientific evidence for the negative effects of feral cats on wildlife especially on islands (Bonnaud et al. 2010, Medina et al. 2011, Nogales et al. 2013). Our study contributes further evidence that feral cat control can increase breeding success and contribute to the population recovery of globally threatened species.

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# Small but fierce: invasive ants kill Barolo Shearwater (*Puffinus lherminieri baroli*) nestling in Cima islet (PORTO SANTO, MADEIRA ARCHIPELAGO)

Pequenas, mas ferozes: formigas invasoras predam cria de Pintaínho (*Puffinus lherminieri baroli*) no ilhéu de Cima  
(PORTO SANTO, ARQUIPÉLAGO DA MADEIRA)

Mário Boieiro<sup>1\*</sup>, Ana Isabel Fagundes<sup>2</sup>,  
Cátia Gouveia<sup>2</sup>, Jaime A. Ramos<sup>3</sup>  
& Dília Menezes<sup>4</sup>

**1** Centre for Ecology, Evolution and Environmental Changes/Azorean Biodiversity Group, Departamento de Ciências e Engenharia do Ambiente, Universidade dos Açores, 9700-042 Angra do Heroísmo, Terceira, Açores, Portugal

**2** Sociedade Portuguesa para o Estudo das Aves, Rua da Mouraria nº9, 4º B - 9000-047 Funchal, Madeira, Portugal

**3** MARE - Marine and Environmental Sciences Centre, Department of Life Sciences, University of Coimbra, 3000-456 Coimbra, Portugal

**4** Instituto das Florestas e da Conservação da Natureza, Quinta do Bom Sucesso, Caminho do Meio, 9050-251 Funchal, Madeira, Portugal

\* Corresponding author: mboieiro@fc.ul.pt



## ABSTRACT

Invasive vertebrate species are known to be a major cause of seabird nestling mortality in oceanic islands. On the other hand, avian nestling predation by invasive ants has seldom been recorded probably due to the confusion of the role of ants as predators or decomposers and the lack of specific studies on this subject. Here, we report the predation of a Barolo Shearwater nestling *Puffinus lherminieri baroli* by the invasive Argentine Ant *Linepithema humile* in Cima islet (Porto Santo, Madeira archipelago). Ants were found in very high abundance in this islet and the recorded attack started immediately as the nestling was hatching. Within a few hours the nestling was overwhelmed by the ants and was dead the next day. Our finding aims to call attention for an overlooked cause of seabird nestling mortality that may be important in areas heavily infested by invasive ants, particularly in island ecosystems.

**Keywords:** biological invasions, crevice-nesting seabirds, *Linepithema humile*, nestling predation, oceanic islands.

## RESUMO

As espécies invasoras de vertebrados constituem uma das principais causas de mortalidade de crias de aves em ilhas oceânicas. Por outro lado, pouco se conhece sobre os impactos das formigas invasoras nas crias de aves, uma vez que o seu papel ecológico de predador ou decompositor é dificilmente distinguido, mas também pela escassez de estudos sobre esta temática.

Aqui, documentamos a predação de uma cria de Pintaíno *Puffinus lherminieri baroli* pela Formiga-argentina *Linepithema humile* no ilhéu de Cima (Porto Santo, Arquipélago da Madeira). Estas formigas invasoras ocorrem em elevada abundância neste ilhéu e o ataque à cria começou imediatamente após a eclosão. A cria foi subjugada pelas formigas em poucas horas, tendo a sua morte sido confirmada no dia seguinte. Com este registo pretendemos destacar uma causa de mortalidade de crias de aves marinhas ainda pouco conhecida, mas que poderá ser importante em áreas infestadas por formigas invasoras, particularmente em ecossistemas insulares.

**Palavras-chave:** invasões biológicas, aves marinhas que nidificam em fendas rochosas, *Linepithema humile*, predação de crias, ilhas oceânicas.

## Introduction

Islands are important breeding locations for many seabird species and play a major role on their conservation at global level (Mulder et al. 2011). During the last centuries, human activities have intensified dramatically at a global scale even reaching the most remote oceanic islands. Consequently, the native island biotas have experienced both direct and indirect effects of human activities, which invariably led to native species declines and extinctions (Blackburn et al. 2004, Wood et al. 2017).

A major cause of the marked decline of seabird populations in islands is the presence of invasive species, particularly cats, rats and mice that prey upon eggs, nestlings and even adult birds (Blackburn et al. 2004, Jones et al. 2008, Towns et al. 2012). The severe impacts of invasive vertebrate species on seabirds breeding in islands are well-documented and are being mitigated all over the world through population control and eradication initiatives (Veitch et al. 2011, Dunlevy et al. 2012, DIISE 2015, Jones et al. 2016). In contrast, there is little information on the impact of invasive invertebrates on seabird popula-

tions (Plentovich et al. 2018). Here we report the predation of a Barolo Shearwater *Puffinus lherminieri baroli* nestling by invasive Argentine ants *Linepithema humile* at Cima islet (Porto Santo, Arquipélago da Madeira). This finding and the recent observations made in Desertas Islands (Boieiro et al. 2018) are to our knowledge the first records of avian nestling predation by invasive ants in Europe.

## Method

The observation of the ant attack to the nestling was made on 9 April 2011 at the southwest side of Cima islet (33° 03' N, 16° 17' W), when we were developing surveys to locate nest sites. The visits to Cima Islet were conducted from 6 to 10 April coinciding with the beginning of the hatching period. During 2011, only 8 nests with nestlings of Barolo Shearwater were found.

The Cima islet has a small surface area (0.32 km<sup>2</sup>) mostly covered by coastal vegetation dominated by *Artemisia argentea* or *Mesembryanthemum spp.*, and is distanced

450 m from the main island of Porto Santo (Madeira archipelago; Alves et al. 2015). It is included in the recently created Network of Marine Protected Areas of Porto Santo, being legally protected since 2008. Thenceforth, it has been the target of several restoration activities to eradicate invasive species (e.g. mice, rabbits and several plant species), recover natural habitats and promote the conservation of threatened species, such as seabirds and endemic snails (Alves et al. 2015). This islet holds small breeding populations of Cory's Shearwater *Calonectris borealis*, Bulwer's Petrel *Bulweria bulwerii*, Band-rumped Storm Petrel *Hydrobates castro* and Barolo Shearwater (Fig. 1).

Barolo Shearwaters nest in natural crevices in dense rubble composed of small to medium (10-100cm diameter) sized rocks (Fagundes et al. 2016). On Cima islet this habitat is found along a narrow strip at the base of cliffs mainly along the southwestern shore. The Barolo Shearwater population for Cima islet is estimated at 129-148 breeding pairs (I. Fagundes, unpublished data).

## Results

During the survey to the Barolo Shearwater colony of Cima islet, we noticed high activity of Argentine ants near the nesting

Figure 1 - Barolo Shearwater (*Puffinus lherminieri baroli*) in its nest at Cima islet.

Figura 1 - Pintaíinho (*Puffinus lherminieri baroli*) no seu ninho no ilhéu de Cima.





sites. The ants were foraging at the soil surface level, moving along trails or randomly searching for food items. In one nest of Barolo Shearwater, we observed many ants moving over and around an egg that was starting to hatch. During the hatching process, the ants began feeding on the fluids that were released from the egg and later were observed entering the egg. Ant activity kept increasing until the nestling was overwhelmed by a swarm of ants that led to its death, probably as a result from the combination of biting and suffocation. The next day we confirmed that the nestling had been killed by the invasive ants. The nestling was found still inside the egg and was

covered with ants feeding on its remains. During our stay in Cima islet we monitored other nests with recently hatched nestlings, but no further ant attacks were observed. We found that there were previous reports of ants disturbing (Fig. 2) and feeding upon Yellow-legged Gull *Larus michabellis* nestlings at Cima islet, but in these latter cases, ants were assumed to be acting as scavengers (I. Silva, pers. comm.). In fact, to be sure that Argentine ants are able to prey upon seabird nestlings, the killing process must be recorded given that ants may play the dual role of predator and scavenger. So, unless specific work is conducted, it will be difficult to document this behaviour as predatory.

Figure 2 - Argentine Ants attacking a nestling of Yellow-legged Gull (*Larus michabellis*).

Figura 2 - Ataque de formigas-argentina a uma cria de Gaivota de patas amarelas (*Larus michabellis*).



## Discussion

The lack of previous records of avian nestling predation by the Argentine Ant in Madeira archipelago is unexpected, considering that this invasive ant was introduced in the mid-18th century (Wetterer et al. 2006, Queiroz & Alves 2016). However, the isolation of the islets where this species is highly abundant, the cryptic nesting habits of seabirds (e.g. crevice-nesting species) and the apparently sporadic nature of the ant attacks may have contributed to its late documentation. There are only a few other worldwide reports of Argentine Ant predation on avian nestlings, but they confirm the generalist feeding behaviour of this invasive ant: the ants may prey upon geese (*Branta sandvicensis*), petrels (*Bulweria bulweri*), gulls (*Larus michahellis*), terns (*Sterna antillarum*, *S. hirundo*) and passerines (*Junco hyemalis*, *Polioptila melanura*, *Serinus canaria*; Krushelnycky et al. 2001, 2005, Suarez et al. 2005, Boieiro et al. 2018).

The drivers of Argentine Ant attacks to avian nestlings remain unclear. This invasive species is found in very high abundance at Cima islet when compared to the main island of Porto Santo (over 30 times more abundant in the former; Boieiro, unpublished data from pitfall trapping), probably benefiting from low competition and predation pressure on the simplified islet ecosystem. However, as suggested by other authors (e.g. Plentovich et al. 2009), the key factor that seems to trigger the attack of several invasive ant species to avian nestlings may be related with the variation in environmental factors and/or food resource depletion. Further studies targeting the interactions between ants and seabirds are needed to identify the drivers of nestling predation by ants and their impact on avian reproductive success. These studies are more important when we consider that Barolo Shearwater is among the less known seabirds in the North Atlantic and the data from the authors suggests a decline on the Madeira archipelago

population due to several factors, including environmental conditions at sea, lack of suitable breeding habitat and nest-site competition with Bulwer's Petrels and Cory's Shearwaters (Fagundes et al. 2016).

During the last decade, there have been several reports on the direct impact of invasive ants, particularly the Yellow Crazy Ant *Anoplolepis gracilipes* and Fire ants *Solenopsis* spp, on avian nestling condition and survival in a few oceanic islands (Matsui et al. 2009, Plentovich et al. 2009, 2011, 2018, DeFisher & Bonter 2013). These studies were very important by highlighting the negative impacts of invasive ants on nestling growth and survival, also providing evidence of seabird population decline in the infested areas. Thus, it became evident that seabird conservation in oceanic islands should consider the monitoring of invasive ant species impacts on avian reproductive success and the need to make decisions concerning their control.

The findings from Madeira archipelago, the first on European territory, show that the impact of invasive ants on seabird nestlings may be a more widespread phenomenon than previously thought. Given the difficulty in detecting ant aggressive behaviours, their impact on seabird reproductive success remains underestimated throughout the world.

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