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1 Habitat management favouring hunted waterbird species prevents distribution  
2 changes in response to climate warming

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31

32

33 **Abstract**

34 Climate warming is driving species to shift their geographic distribution poleward to track suitable  
35 climatic conditions. Two strategies have been suggested to help species respond to climate warming:  
36 facilitating distribution change or improving persistence. We questioned whether habitat  
37 management in favour of duck hunting activities interacted with duck responses to climate warming.  
38 We studied non-breeding waterbird community changes (110 species) over 28 years at 851 sites in  
39 South-West Europe, where habitat management is a common practice to attract and hunt ducks. We  
40 hypothesized that duck species targeted by habitat management do not need to track temperature  
41 changes as much as non-hunted species, because of the availability of suitable habitats provided by  
42 hunting land managers. We used the Community Temperature Index (CTI) to assess the temporal  
43 responses of communities and species to climate warming. We compared the effect of hunting status  
44 to other functional traits on species responses, controlling for phylogenetic relatedness. The CTI trend  
45 has increased over the study period, indicating a community adjustment to climate warming. However,  
46 hunted ducks contributed to almost 40% of the negative contributions to this community adjustment,  
47 suggesting that hunted ducks do not shift their distribution as much as the other waterbirds do. Winter  
48 fidelity associated with the provision of attractive feeding grounds might explain why ducks did not  
49 seem to shift their distribution in response to climate warming. This study suggests the broad impact  
50 of human activities on wildlife, including on large-scale distribution processes, and questions the long-  
51 term consequences on duck populations.

52 **Keywords:** climate change, duck, Community Temperature Index, Mediterranean, persistence,  
53 wintering

## 54 **Introduction**

55 Global warming is a major threat to biodiversity, leading to massive species distribution changes  
56 (Parmesan & Yohe 2003). Global temperatures have increased by 1.1°C since the pre-industrial period  
57 (IPCC 2021), and are expected to reach +2°C to +5°C by 2100 (Raftery et al. 2017). A possible step  
58 backwards to pre-industrial temperatures in the next centuries is uncertain (IPCC 2021). Therefore,  
59 temperature changes are not only rapid, but also likely to last over time.

60 Species responses to climate warming are diverse but their intensity is largely insufficient (Radchuk et  
61 al. 2019, Lenoir et al. 2020). Species may respond to climate warming through changes in migration  
62 distance, geographic shift, phenology, plasticity, or microevolution (Jiguet et al. 2007, Moller et al.  
63 2010, Dawson et al. 2011, Elmer et al. 2014). The most commonly observed and documented  
64 response is geographical distribution change, leading to changes in community composition in  
65 response to temperature changes at the local scale (Devictor et al. 2012). For instance, temperature  
66 increases might cause local population increases at the cold edge of species distributions, and  
67 decreases in population sizes at the warm distribution edge (Parmesan & Yohe 2003, Thomas et al.  
68 2006, Pavón-Jordan et al. 2015). Species can also persist locally, either by finding micro-refuges (e.g.  
69 at higher altitude, Massimino et al. 2020) or by tolerating changes through phenotypic or genetic  
70 adaptation (plasticity, micro-evolution) (Charmantier et al. 2008, Karell et al. 2011). Functional traits  
71 play an important role to shape species responses to climate warming (Brommer 2008). Species  
72 sharing the same characteristics, like climatic niche, body mass, migration strategies, might be more  
73 likely to respond in a similar way (Jiguet et al. 2007) while facing the same vulnerabilities to  
74 temperature increase (Aubin et al. 2016). Species responses to climate warming may also be  
75 modulated by local environmental factors, like habitat quality (Dawson et al. 2011, Lenoir et al. 2020)  
76 and conservation efforts (Gaget et al. 2022).

77 Helping species respond to climate warming is mainly considered under two complementary climate  
78 adaptation strategies: facilitation and persistence (Bonebrake et al. 2018, Prober et al. 2019, Pearce-  
79 Higgins et al. 2022). Facilitation focuses on helping species shifting their geographical distribution and  
80 settle in new areas in response to global warming (e.g. Maclean & Rehfish 2008, Guillemain & Hearn  
81 2017). It aims at adjusting the rate of species distribution change to the thermal isocline shift. For  
82 example, increasing connectivity between protected habitats facilitates colonization processes in  
83 response to climate warming (McGuire et al. 2016). Conversely, persistence aims at maintaining  
84 species in an environment despite temperature changes. It may indirectly be based on the intrinsic  
85 abilities of species to tolerate the changes (e.g. plasticity) or involve management measures to provide  
86 additional resources or create microrefugia benefiting a targeted species. For example, food supply  
87 may decrease the local extinction risk of species vulnerable to climate warming (Correia et al. 2015).  
88 Both facilitation and persistence strategies are likely to influence the response of species to climate  
89 warming (Bonebrake et al. 2018, Prober et al. 2019).

90 Empirical evidence of long-term species response to climate change has rarely been demonstrated  
91 under the persistence strategy (Correia et al. 2015, Pearce-Higgins et al. 2019). Indeed, as  
92 anthropogenic climate change is a relatively recent biodiversity pressure, conservation measures have  
93 not primarily aimed at improving long-term species persistence in response to temperature increase  
94 specifically. In contrast, efforts have historically aimed at improving species conservation, with  
95 consequences on their distribution regarding temperature change. For example, the creation of

96 protected areas to preserve migratory waterbirds during cold spells in Western Europe (Ogilvie 1983,  
97 Guillemain et al. 2002) are also buffering the negative impact of temperature increase on some cold-  
98 dwelling waterbirds (Gaget et al. 2021). Similarly, the provision of supplementary foods in British  
99 gardens changed the migration behaviour of the Eurasian blackcap (*Sylvia atricapilla*), of which many  
100 individuals are no longer migrating South (Plummer et al. 2015). Such anthropogenic influences can be  
101 examined a posteriori in the light of climate warming (Pearce-Higgins et al. 2022).

102 Here, we question the effect of habitat management for hunting purposes on waterfowl responses to  
103 climate warming at international scale in South-West Europe. Duck hunting is a popular practice in this  
104 region yielding 2.4 million harvested ducks annually (EEA 2020). Indeed, habitat management for  
105 hunting purposes is a local, but widespread practice (FACE 2019), aiming at attracting targeted  
106 waterfowl species to hunting grounds during the non-breeding period. Pro-active habitat management  
107 actions for hunting purposes include the creation of man-made ponds, spreading of seeds as bait,  
108 artificial flooding, vegetation clearing including the creation of openings in reedbeds, and other actions  
109 (e.g. Mathevet & Guillemain 2016, Vallecillo et al. 2019). Economic incentive for such intensive  
110 management actions is that sites with high abundance of ducks can yield high income by renting out  
111 hunting rights (e.g. Niang et al. 2016). Specific habitat management targeted to boost abundance of  
112 hunted species is often carried out in large private properties, e.g. in the Camargue, Southern France  
113 (Tamisier & Grillas 1994) and provide prestigious social status and reputation of the land-owner, e.g.  
114 in Italy. However, if such management measures attract hunted duck species in large numbers, they  
115 may not be so attractive to other waterbird species, unless for species sharing common habitat or diet  
116 characteristics with hunted ducks, such as granivorous and herbivorous species (Tamisier and Dehorter  
117 1999, Mouronval et al. 2014). In particular, such habitat management are not expected to provide  
118 benefits to invertebrate or fish eating species (e.g. shorebirds, herons) (Tourenq et al. 2000). Gaget et  
119 al. (2018) showed that in the south-western European Union Member States, waterbird responses to  
120 climate warming were correlated with their protection status. Hunted species had a lower response to  
121 climate warming through distribution change than the other species. Here, we further explore the  
122 reason for this interaction between hunting status and climate warming, by investigating the possible  
123 role of habitat management practices on hunted ducks.

124 We studied community reshufflings from 110 non-breeding waterbird species in south-western Europe  
125 over 28 years on 851 sites. Many of these waterbirds breed at northern latitudes and overwinter in  
126 wetlands in this region. Our main interest was to explore whether the distributional changes of duck  
127 species, which are actively targeted by hunting in this region, differ from those of other hunted and  
128 non-hunted waterbird species, while controlling for functional traits that differ between these species.  
129 To infer distribution changes we used the Community Temperature Index (CTI) framework, a  
130 community weighted mean index used to measure the average temperature affinity of all the species  
131 of a community, based on individual species temperature index (STI) (Devictor et al. 2012). We  
132 assessed temporal changes of CTI over years for the non-breeding waterbird community using  
133 abundance time series collected at site level through the International Waterbird Census (Delany 2010)  
134 and STI calculated as the mean winter temperature values across the non-breeding distribution of the  
135 species (Gaget et al. 2021). Under a climate warming scenario, we expected the avian community at  
136 a given site to change over time towards species with a higher thermal affinity. Thus, the CTI was  
137 expected to increase over time. Such community reshuffling has already been documented for non-  
138 breeding waterbirds in the western Palearctic (Gaget et al. 2021) and in numerous other studies  
139 (Devictor et al. 2012, Auffret and Thomas 2019, Fourcade et al. 2019). We then assessed the species'

140 contributions to this temporal increase in CTI. Our main interest was whether this contribution  
141 depends on the species' hunting status (hunted ducks, other hunted species and non-hunted species),  
142 while accounting for functional traits (species thermal affinity, body mass, generation length, diet and  
143 migration strategy) and phylogeny (Jiguet et al. 2007, Brommer 2008) to control for potential  
144 confounding effects. Because of the intensive management practices in the study region (Tamisier &  
145 Grillas 1994, Pernellet et al. 2015a, Zenatello et al. 2021), we expected that hunted ducks would be  
146 slower to adjust their distribution to climate warming compared to non-hunted waterbirds, because  
147 management of hunting areas may provoke a greater site fidelity. We thus expected that hunting  
148 status shapes a species' response to climate warming.

149

## 150 **Methods**

151 We studied non-breeding waterbird abundance time series collected from 1993 to 2020 at 851 sites in  
152 Portugal, Spain, France and Italy. The data were collected for the International Waterbird Census  
153 coordinated by Wetlands International (Delany 2010), resulting in one survey per year performed by  
154 skilled ornithologists in mid-January on hundreds of wetlands. We filtered the original dataset ( $n =$   
155 3,288 sites) to ensure the quality of the data. To do so, we restricted the list of waterbird species to  
156 the list of the Agreement on the Conservation of African-Eurasian Migratory Waterbirds ([www.unep-  
158 aewa.org](http://www.unep-<br/>157 aewa.org)), we removed count events with less than two species to allow estimation of species  
159 contributions to CTI trend (see below) and we removed species with less than 100 individuals observed  
160 among all sites included in the study. Because of the particular focus on ducks, we removed sites  
161 without any hunted duck observation. We retained sites with at least five surveys and one survey per  
162 decade (1990s, 2000s and 2010s), resulting in a subset of 851 sites with  $20.5 \pm 6.2$  surveys on average  
163 ( $\pm$ SD) (see map Appendix 1). Considering taxonomic changes that occurred during the study period,  
164 *Larus michahellis*, *L. cachinnans* and *L. argentatus* were lumped into one 'species' *L. michahellis*. This  
165 lumping has a negligible impact on most of the study area where *L. michahellis* is largely dominant over  
166 the two other species, with the exception of the French Atlantic coast. The dataset used for the  
167 analyses resulted in 17,462 surveys of 851 sites over 28 years, with a cumulative record of 119 million  
168 birds from 110 species (Appendix 2).

168 We considered three hunting status categories between species: eleven duck species mainly targeted  
169 by habitat management, 25 other hunted waterbirds and 75 non-hunted species (Appendix 2). Hunted  
170 ducks are Northern Pintail (*Anas acuta*), Common Teal (*A. crecca*), Mallard (*A. platyrhynchos*), Eurasian  
171 Wigeon (*Mareca penelope*), Gadwall (*M. strepera*), Northern Shoveler (*Spatula clypeata*), Garganey (*S.  
172 querquedula*), Common Pochard (*Aythya ferina*), Tufted Duck (*A. fuligula*), Greater Scaup (*A. marila*)  
173 and Red-crested Pochard (*Netta rufina*). These species are all listed in Annex II of the Birds Directive,  
174 meaning that they can be hunted according to the regulation of the European Union or Member States  
175 (Red-crested Pochard is hunted in France and Spain only, and Greater Scaup is hunted in France only).  
176 The other hunted species are listed in Annex II of the Birds Directive and hunted at least in one Member  
177 State of the study area, and include species such as Greylag Goose (*Anser anser*), Coot (*Fulica atra*),  
178 Lapwing (*Vanellus vanellus*) and Eurasian Curlew (*Numenius arquata*). The non-hunted species are not  
179 listed in Annex II, or listed in Annex II but not hunted in the study area, including species such as Mute  
180 Swan (*Cygnus olor*), Ferruginous Duck (*Aythya nyroca*), Grey Heron (*Ardea cinerea*), Eurasian Spoonbill

181 (*Platalea leucorodia*), Little Grebe (*Tachybaptus ruficollis*), Great Cormorant (*Phalacrocorax carbo*) and  
182 Mediterranean Gull (*Larus melanocephalus*).

183 We document species functional traits and phylogenetic proximities to control for potential  
184 confounding effects (Appendix 2). Functional traits were species thermal affinity, body mass,  
185 generation length, diet and migration strategy. These functional traits are expected to influence bird  
186 response to climate warming (Jiguet et al. 2007, Brommer 2008). Species thermal affinity was defined  
187 as the Species Temperature Index (STI), calculated as the average temperature in January (1950-2000,  
188 [www.worldclim.org](http://www.worldclim.org), spatial resolution of 0.25°) on the non-breeding species distribution (BirdLife  
189 International & HBW 2018) from Gaget et al. (2018). Body mass (continuous, log transformation) and  
190 diet (Plant, Omnivore, Invertebrate, Vertebrate) were extracted from Wilman et al. (2014) (where  
191 omnivore corresponds to a combination of less than 50% of all other categories), generation length  
192 (continuous) from IUCN (2021) and migration strategy (resident, short, long) from Storchová et al.  
193 (2018). Migration strategies were represented in similar proportions between hunting status  
194 categories, with most of hunted ducks being short to long-distance migrants (Storchová et al. 2018;  
195 Appendix 2). To control for species phylogeny, we used phylogeny trees from BirdTree  
196 (<https://birdtree.org>).

197 We used temperature anomalies to assess changes in winter temperatures on the survey sites over  
198 the study period. We collected average monthly temperatures gridded at 0.5° resolution, from the  
199 Climatic Research Unit v4.05 (Harris et al. 2020). Winter temperatures anomalies were computed each  
200 winter at site level as the average of the mean monthly temperatures for the 3-month period of  
201 November, December and January, i.e. the duration expected to impact waterbird distribution at  
202 winter time (Gaget et al. 2020), minus the average monthly temperatures from November to January  
203 for the 1993-2020 period.

204

## 205 **Statistical analyses**

206 To assess and explain patterns of species responses to climate warming, we first calculated the long-  
207 term trend of CTI over years, then we assessed species contribution to the CTI trend, and finally we  
208 investigated the effect of hunting status and functional traits on species contribution, taking species  
209 phylogeny into account.

210 The CTI is a community weighted mean index used to measure the average temperature affinity of all  
211 the species included in the community based on individual STI (Devictor et al. 2012). The CTI is the  
212 average STI weighted by the  $\log_e(\text{abundance} + 1)$  of the species observed during a survey (Gaget et al.  
213 2018). We calculated one CTI value per count event, based on all the species observed at this time  
214 among the 110 waterbird species included in the study. Over time, a CTI can increase in response to  
215 climate warming by a relative increase of species with high STI, or a relative decrease of species with  
216 low STI.

217 We assessed CTI temporal trends with a linear mixed effect model (LMM) with the CTI as the response  
218 variable and year as a fixed effect. Sites, nested in countries, were used as a random effect to account  
219 for heterogeneity among sites and national monitoring schemes (Gaget et al. 2018). Spatial  
220 autocorrelation was taken into account with an exponential structure on the geographic coordinates

221 (Gaget et al. 2018). We used the same model with temperature anomalies as response variable, to  
222 assess temporal changes in temperature anomalies. To visualize inter-annual changes in CTI and  
223 temperature anomalies, we also used the same models but with year as a categorical variable.

224 To assess the contribution of each species to CTI trends, we ran a jackknife analysis by removing each  
225 species in turn from the CTI before running LMMs on partial CTIs. Hence, species contribution to the  
226 CTI temporal trends was assessed as the difference between the original temporal trend (all species  
227 included) and the temporal trend of the model based on the partial CTI (all but one species). Species  
228 contributions were then reported as a percentage of influence, positive or negative, on the CTI trends  
229 (Gaget et al. 2018). Note that species contributions are relative values, comparing species to the whole  
230 community. A negative contribution to a positive CTI trend means that a species is contributing  
231 relatively less than the other species to the community adjustment to climate warming.

232 Finally, we used phylogenetic generalized least squares (PGLS) models to assess differences between  
233 species contribution to CTI changes. We compared species contributions between hunting statuses  
234 and functional traits (species thermal affinity, body mass, generation length, diet and migration  
235 strategy), considering species phylogeny. We used a forward model selection approach to select the  
236 most parsimonious model with predictors explaining the most species contributions, i.e. selecting  
237 models with the lowest AIC among a  $\Delta AIC < 2$  (Burnham and Anderson 2002). Phylogeny is a  
238 potential factor structuring adaptive bird response to climate change (Hällfors et al. 2020). Hunted  
239 ducks are from the same Anatidae family and often share the same genus. Their intrinsic sensitivity to  
240 climate warming and adaptive capacities are likely closer among each other than with species from  
241 other taxonomic clades. Consequently, contrasting species groups exposes the analytical framework  
242 to a potential confounding effect. To avoid this bias, we accounted for phylogenetic relatedness by  
243 adding the phylogeny tree as a Brownian correlation structure in the models. In addition, we assessed  
244 the multicollinearity between functional traits with the Variance Inflation Factor (VIF).

245 We performed two sensitivity analyses (Appendix 3) to ensure that the results were independent of 1)  
246 the site protection status, because protected areas can also interact with wintering waterbird  
247 responses to climate warming (Gaget et al. 2021), and of 2) the number of surveys per site, by selecting  
248 only sites with more than 25 surveys over the 28 years period.

249 Statistical analyses were performed in R 4.1.2 (R Development Core Team 2021). LMM and PGLS were  
250 run with package 'nlme' (Pinheiro et al. 2017), Nagelkerke pseudo R-squared were assessed with  
251 package 'rcompanion' (Mangiafico 2016) and VIF with package 'performance' (Lüdecke et al. 2019).

252

## 253 **Results**

254 The Community Temperature Index (CTI) showed a very clear linear increase over years ( $\beta = 0.045 \pm$   
255  $0.001$  SE,  $p < 0.001$ ), indicating that the avian community was, on average, changing according to  
256 species thermal affinity (Fig. 1). Winter temperatures increased by  $0.6^\circ\text{C}$  on average over the study  
257 period on the studied sites ( $\beta = 0.023 \pm 0.001$  SE,  $p < 0.001$ ).

258 However, not all species contributed equally to this change in CTI. Species diet and hunting status were  
259 the main factors explaining species contribution ( $\beta_{\text{Plant}} = -2.61 \pm 1.19$  ( $\pm$ SE)  $[-4.74; -0.48]$  (95%CI),  
260  $\beta_{\text{Invertebrate}} = -0.15 \pm 1.19$   $[-0.39; 0.09]$ ,  $\beta_{\text{Vertebrate}} = -1.27 \pm 1.20$   $[-1.45; -1.09]$ ,  $\beta_{\text{Omnivore}} = 1.06 \pm 1.19$

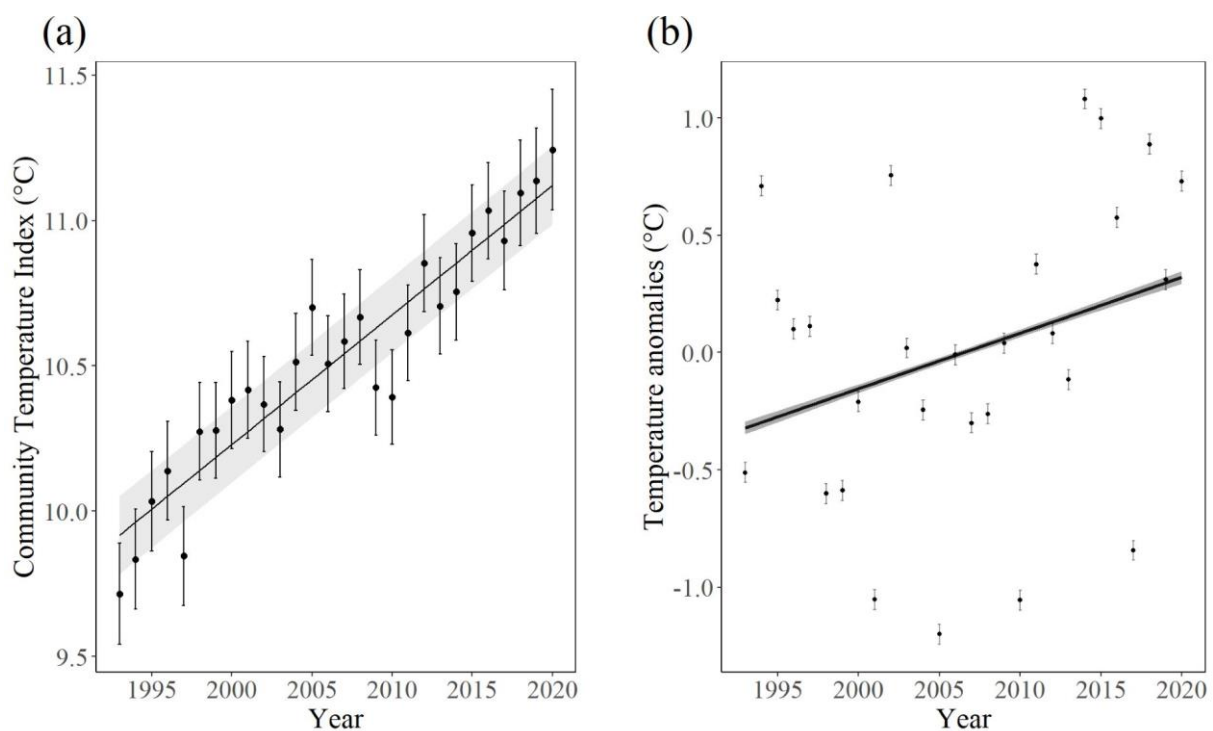


261 [0.86;1.26];  $\beta_{\text{Hunted ducks}} = -3.62 \pm 1.24 [-5.09;-2.15]$ ,  $\beta_{\text{Other hunted}} = 0.57 \pm 1.19 [-2.20;1.57]$ ,  $\beta_{\text{Non-hunted}} = 0.82$   
 262  $\pm 1.19 [0.15;0.89]$ ), whereas other predictors had little effect (Table 1, Appendix 4). Multicollinearity  
 263 between species diet and hunting status was low (VIF = 1.05). Most of the variance explained by the  
 264 model was related to species diet (82%), with plant eating species contributing predominantly  
 265 negatively to the CTI change (Fig. 2). Indeed, 83% of the plant-eating species had negative  
 266 contributions, compared to 48%, 44% and 64% for species eating invertebrates, vertebrates and being  
 267 omnivores, respectively (omnivores potentially comprising some plant-eating diet). Hunting status also  
 268 strongly affected species contributions to CTI change (Table 1, Fig. 2). Most (73%) of hunted duck  
 269 species contributions were negative (summing to  $-39.8\%$ ). Only Mallard, Greater Scaup and Garganey,  
 270 among 11 hunted duck species, made positive contributions to CTI change (summing to  $+15.5\%$ ),  
 271 although the Mallard accounted for almost all positive contribution ( $+14.9\%$ ; Fig. 2). Among the other  
 272 hunted and non-hunted species, 63% and 52% of the species contributed negatively, respectively.

273 The sensitivity analyses conducted 1) without sites included inside protected areas and 2) only on site  
 274 with more than 25 surveys, returned fairly similar results, indicating the importance of the species  
 275 hunting status to explain species contributions to CTI change (Appendix 3).

276

277

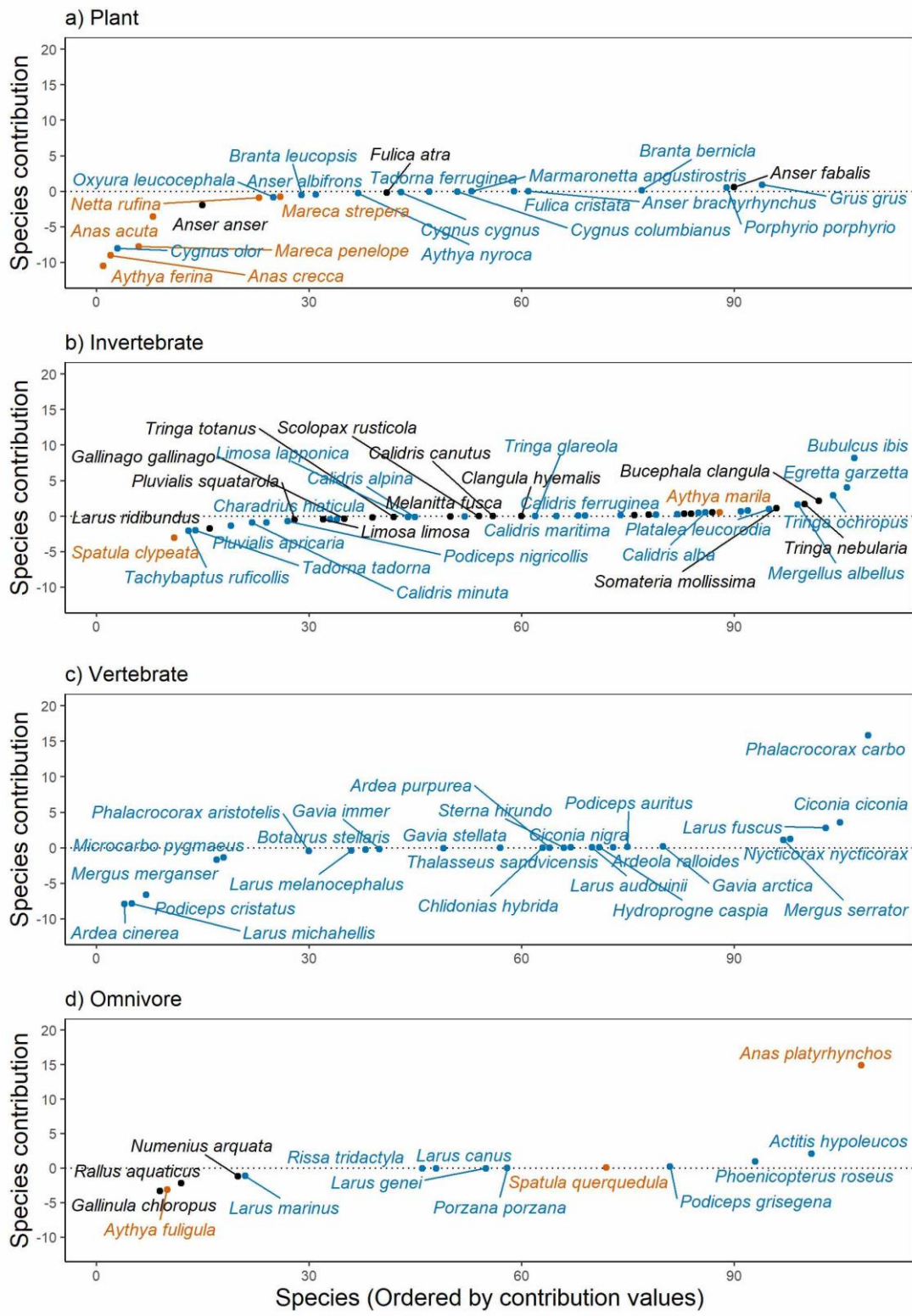


278

279 Figure 1: Temporal trends of (a) Community Temperature Index (CTI)  $\pm 95\%$  CI and (b) temperature  
 280 anomalies  $\pm 95\%$  CI. Note that although temperature and CTI share the same unit, the scales are not  
 281 proportional, i.e. a temperature increase of  $+1^\circ\text{C}$  does not lead to the same CTI increase (Gaget et al.  
 282 2021).

283

284



289 Figure 2: Species contributions (%) to the Community Temperature Index (CTI) temporal trend,  
 290 depending on species diet (plant, invertebrate, vertebrate, omnivore) and hunting status (hunted  
 291 ducks, other hunted and non-hunted), with species ordered by their contribution value (n = 110  
 292 species). The horizontal dotted line represents a null contribution. Not all species names are shown for  
 293 aesthetic reason.

294

295 Table 1: Selection of the best models based on AIC and parsimony (Delta AIC <2 and lowest DF),  
 296 identifying predictors the most related to species contributions. The F-value corresponds to the sum  
 297 of the effects and DF is degrees of freedom. The best model to explain species contribution considers  
 298 both species hunting status and diet (in bold). Models are fitted with phylogenetic generalized least  
 299 squares (PGLS). Diet, hunting status and migration are factors and STI (Species Temperature Index),  
 300 body mass and generation length are quantitative variables (See Appendix 4 for full model selection).

Formula	AIC	Delta AIC	Pseudo R-square	F-value	DF
<b>Diet + Hunting status</b>	<b>697.2</b>	<b>1.4</b>	<b>0.058</b>	<b>2.329</b>	<b>6</b>
Hunting status + Diet + Generation length	695.8	0.0	0.097	6.835	7
Hunting status + Diet + Body mass	696.3	0.5	0.077	4.410	7
Hunting status + Diet + Migration	696.5	0.7	0.060	2.410	8

301

302

### 303 Discussion

304 Over the study period (1993-2020), the observed CTI increase demonstrates that non-breeding  
 305 waterbird communities have changed, with warm-dwelling species becoming relatively more  
 306 abundant than cold-dwelling species in the region. This community change was expected because of  
 307 the temperature increase in the study area, suggesting that non-breeding waterbird communities are  
 308 reshuffling in response to climate warming. This finding is fully consistent with other studies on non-  
 309 breeding waterbirds (Godet et al. 2011, Pavón-Jordán et al. 2017, Gaget et al. 2018, Gaget et al. 2021),  
 310 breeding birds (Devictor et al. 2012) and other taxa (Devictor et al. 2012, Auffret and Thomas 2019,  
 311 Fourcade et al. 2019) in Europe. This reshuffling is a long-term process caused by species distribution  
 312 changes in response to climate warming, with warm-dwelling species colonizing new sites at the  
 313 leading distribution edge and cold-dwelling getting extirpated at the trailing distribution edge (Thomas  
 314 et al. 2006, Gaget et al. 2021).

315 While the community on average has responded to climate change, individual species responses  
 316 varied. Some species distribution strongly tracked climate warming, and such species had, in the CTI  
 317 framework used in this study, positive contributions to the CTI trend. Examples include Common  
 318 Sandpiper (*Actitis hypoleucos*), Goldeneye (*Bucephala clangula*) and White stork (*Ciconia ciconia*).  
 319 However, other species barely adjusted their distribution to climate warming and these species had  
 320 negative contributions to the CTI trend. We here showed that most ducks targeted by habitat  
 321 management for hunting purposes (except Mallard, and to a lesser extent Garganey and Greater  
 322 Scaup) had negative contributions to the CTI trend. This finding therefore suggested that habitat  
 323 management may influence species response to climate warming. We believe there are two non-  
 324 mutually exclusive explanations for this finding.

325 Firstly, managing sites to benefit hunted species may have lowered the propensity of these species to  
326 adjust their distribution to climate warming. Indeed, intensification of habitat management for hunting  
327 purposes has been suggested as one of the reasons for contemporary changes in duck diet, body  
328 condition and migration phenology (Mathevet & Tamisier 2002; Guillemain et al. 2015). Various  
329 habitat management actions are commonly used to attract wintering ducks (e.g. Mathevet &  
330 Guillemain 2016, Vallecillo et al. 2019). One potentially important management action is food  
331 supplementation, where seeds (wheat, corn or rice) are provided as bait, a common practice in South-  
332 West European countries (Brochet et al. 2012). For example, such bait may account for half of the Teal  
333 food intake in some areas (Legagneux 2007), and feeding activity for this duck species has shifted these  
334 past decades from natural habitats to artificially managed freshwater marshes (Brochet et al. 2012).  
335 Over the same period, Teal body mass has increased by more than 10% at their major Camargue  
336 wintering site in southern France (Guillemain et al. 2010), and arrival date at this wintering ground has  
337 advanced (Guillemain et al. 2015). Intensive water level management is another management practice  
338 likely influencing duck distribution change or local persistence in response to climate warning.  
339 Wetlands are flooded in summer, including temporary ponds that otherwise would be dry. The aim of  
340 this summer flooding is to maximize plant biomass growth in these ponds to coincide with the arrival  
341 of ducks from their breeding grounds in autumn (Tamisier & Grillas 1994, Davis et al. 2014).  
342 Supplementation of freshwater in brackish habitats (e.g., coastal lagoons), aims also to increase habitat  
343 diversity hence attracting freshwater duck species. All such management actions improve habitat  
344 quality and attractiveness to ducks, resulting in large alterations of duck distribution in the last  
345 decades, which now concentrate in areas of intensive habitat management for hunting purposes  
346 (Zenatello et al. 2014, 2021). Habitat management for hunting purposes may hence have counter-  
347 balanced the effect of climate warming and “retain” hunting ducks to these wintering grounds, which  
348 they would otherwise abandon for more northern areas.

349 Secondly, hunted ducks may have been slower to adjust their distribution to climate change because  
350 agricultural practices were particularly beneficial for these species. In particular, rice fields are flooded  
351 and used as foraging grounds by ducks during the non-breeding season, with ducks feeding on non-  
352 harvested rice and weed seeds (Pernollet et al. 2015a). Other waterbird species use rice fields for  
353 feeding to a lesser extent, except for species that can specialize on rice, such as the Greater Flamingo  
354 (*Phoenicopterus roseus*) during the seeding period in spring (Pernollet et al. 2015a). In Portugal, Spain,  
355 France and Italy, some wetlands hosting a large number of ducks are close to large rice producing  
356 areas, e.g. Ebro delta, Albufera, Camargue, Pavia. In such regions, the number of wintering ducks was  
357 positively related to the area of flooded harvested ricefields (Pernollet et al. 2015b). Furthermore, rice  
358 farming practices and management of sites to promote hunting are not mutually exclusive, since  
359 harvested ricefields can also be flooded for hunting purposes (Niang et al. 2016, Pernollet 2016).  
360 Indeed, in addition to agronomic benefits, this practice provides suitable wintering grounds for ducks  
361 and may therefore promote site fidelity.

362 Despite these explanations, our current analysis cannot definitely infer the reasons for the slower  
363 distribution adjustment of hunted ducks to climate warming compared to other waterbirds. In  
364 particular, we conducted a comparison between species and not between sites. To establish that  
365 hunting-oriented site management promotes persistence of hunted ducks, site-level information on  
366 management practices and intensity in South-West Europe is needed. Spatialized data on habitat  
367 management for hunting purposes are currently lacking at regional or national scale. Importantly,  
368 ducks are often mobile over large areas within their winter range (Caizergues et al. 2011, Gourlay-

369 Larour et al. 2012) and may hence be affected by large scale management driven by regional factors  
370 in addition to site-specific management, such as land-use policy or socio-cultural practices. The fact  
371 that ducks commute daily between a day-roost (generally protected sites where they can rest and  
372 where they are surveyed) and surrounding nocturnal feeding areas (functional units *sensu* Tamisier &  
373 Tamisier 1981) further inflates uncertainty in how these species are affected by habitat management  
374 in a given area (Vallecillo 2021). An experimental design contrasting the effect of site management on  
375 duck carrying capacity would be challenging because of the mismatch between diurnal bird monitoring  
376 and nocturnal feeding behaviour (Brochet et al. 2009).

377 Responses to climate warming differ between species for many reasons, including, among others,  
378 species movement ability or adaptive capacities, (Dawson et al. 2011). Our best model only explained  
379 5.8% of the differences between species contributions, suggesting that an important part of the  
380 variation was not captured by the tested functional traits. This might have been expected because  
381 waterbirds share more ecological characteristics than we could get by analyzing a more  
382 phylogenetically diverse bird community (e.g. Jiguet et al. 2007, Brommer 2008). Consequently,  
383 accurate traits might be neglected, e.g. at population-level across species flyways, but needed to  
384 reflect local adaptations to environmental parameters, including climate warming (DeMarche et al.  
385 2019).

386 Waterbird diet was an important predictor of species contribution to CTI trend, but other functional  
387 traits were not. Plant eating species mainly contributed negatively to the community changes. One  
388 likely reason might be that species sharing similar feeding ecology to hunted ducks, mostly plant  
389 eaters, were also affected by habitat management, like coots and swans (Brochet et al. 2009). Thus,  
390 habitat management for hunting purposes may also benefit plant eaters in general, to some extent.  
391 Generation length actually explained a fair amount of variation, but was not selected according to  
392 parsimony. It may therefore be an additional factor explaining why some species have a delayed  
393 distribution change (Maclean et al. 2008). Indeed, juvenile settlement can influence over-winter  
394 location when mature, meaning that long-lived species might need more time to shift their distribution  
395 than short-lived species (Godet et al. 2011).

396 Mallards did not follow the same pattern as the other hunted ducks. We found that their distribution  
397 was tracking climate warming. Indeed, this species has started to shorten its migration distance  
398 decades ago along its European flyway, likely in response to temperature increases (Sauter et al. 2010;  
399 Gunnarsson et al. 2012, Guillemain et al. 2015). Previous work highlighted that the Mallard clearly  
400 responds to winter temperatures, both in North America (Schummer et al. 2010) and in Europe (Dalby  
401 et al. 2013). Its metabolism and opportunistic feeding strategy may explain these findings (Schummer  
402 et al. 2010): its larger size and weight allow Mallard to endure negative temperatures more easily than  
403 smaller species, disadvantaged by their greater surface area to volume ratio. It is also a relative habitat  
404 generalist species exploiting numerous wetland habitats, small, even degraded or urban and can switch  
405 from water to terrestrial feeding while the other duck species are more wetland dependent.  
406 Accordingly, Mallard are likely to winter closer to their temperature limits than other duck species,  
407 resulting in a different trade-off between saving energy from migration and thermal regulation, and  
408 likely a quicker response to climate warming by a distribution change. It cannot be excluded, however,  
409 that the fact that millions of mallards are released annually in Europe (Champagnon et al. 2013) also  
410 blurs the relationship between environmental conditions and the geographic distribution of this  
411 species (Söderquist et al. 2013). Garganey and Greater Scaup were the two other duck species with a

412 positive contribution to the CTI trend. This result should however be considered with great caution,  
413 given both species are rare winter visitors in the study area, the former usually wintering in Africa and  
414 the latter at more northern latitudes. Indeed, Garganey and Greater Scaup numbers had a cumulative  
415 abundance of 983 and 37,634 counted individuals in our dataset, respectively, compared to 15.3  
416 million Mallards.

417

## 418 **Perspectives**

419 The economic incentive provided by hunting is expected to drive landowners to manage habitat to  
420 locally boost hunted species abundance. This kind of habitat improvements could locally release  
421 hunted species from some of the adverse effects of climate change, but also impede current or future  
422 distributional changes. Consistent with this prediction, and based on community changes in 110  
423 waterbird species over a large region (south-west Europe), we demonstrated here that hunted ducks  
424 are indeed slower in their response to climate warming compared to other waterbirds. Although our  
425 analysis did not directly infer that site management for hunting purposes drives this outcome, we  
426 suggest the pattern we found may be general. Because hunting is a global phenomenon, more studies  
427 exploring the interactions between site management and climate change as drivers of altered  
428 distributional changes are needed to understand whether hunted ducks are more persistent against  
429 climate change because of habitat management, or trapped there despite the ongoing climate  
430 warming. While hunted ducks may in the short-term benefit from such artificial land management, this  
431 may already have negative consequences for other elements of biodiversity (Tamisier & Grillas 1994),  
432 and may be detrimental for the ducks themselves in the longer run.

433

434 In addition to the potential effect of habitat management on waterbird responses to climate warming,  
435 harvesting can also have a direct impact on population dynamics through removal of individuals (Cooch  
436 et al. 2014). Among studied populations of non-breeding hunted ducks, several were stable or  
437 increasing over both short and long terms, such as Northern Pintail, Common Teal, Gadwall and Red-  
438 crested Pochard, while for others the trends were declining at least in one studied country, such as in  
439 Mallard, Eurasian Wigeon, Common Pochard and Tufted Duck (EEA 2020). Sustainable management of  
440 duck populations is required to maintain these species at a favorable conservation status.  
441 Management planning needs international collaborations at species distribution level, using hunting  
442 data and demographic parameters (Holopainen et al. 2018). Whether the distribution of harvested  
443 populations is changing or not in response to climate warming should be carefully considered, to  
444 anticipate ecologic, cultural and economic challenges (Guillemain et al. 2013).

445

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452

453 **Author Contributions Statement**

454 EG, TG and MG conceived the ideas and designed methodology; NB, TL, BL, FM, CM and MZ collected  
455 the data; EG analysed the data; EG and MG led the writing of the manuscript. All authors contributed  
456 critically to the drafts and gave final approval for publication.

457

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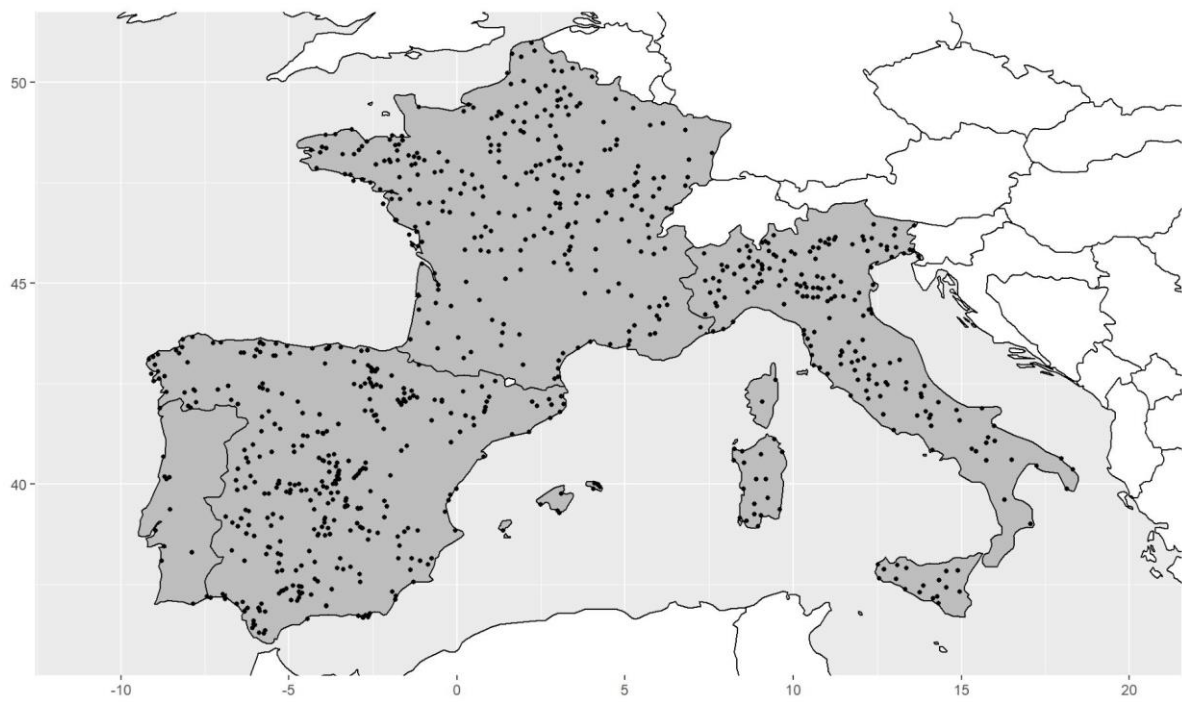
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685 Appendix 1. Map of the study area, with the 851 monitored sites.

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688 Appendix 2. Species functional traits and hunting status. We considered three hunting status  
689 categories between species: eleven duck species listed in Annex II of the Birds Directive, the  
690 other species listed in Annex II of the Birds Directive hunted in at least one Member State, and  
691 the non-hunted species not listed in Annex II, or listed in Annex II but not hunted in the study  
692 area. The STI was calculated as the average temperature in January (1950-2000,  
693 www.worldclim.org, spatial resolution of 0.25°) on the non-breeding species distribution  
694 (BirdLife International & HBW 2018) from Gaget et al. (2018). Body mass (continuous, log  
695 transformation) and diet (Plant, Omnivore, Invertebrate, Vertebrate) were extracted from  
696 Wilman et al. (2014), generation length (continuous, in years) from IUCN (2021) and migration  
697 strategy (resident, short, long) from Storchová et al. (2018).  
698

Species	Diet	Hunting status	Generation length	STI	Migration strategy	Body mass (log)
<i>Oxyura leucocephala</i>	Plant	Non-hunted	6	1.26971416	Short	6.50
<i>Clangula hyemalis</i>	Invertebrate	Other hunted	9	-2.499599758	Short	6.77
<i>Somateria mollissima</i>	Invertebrate	Other hunted	9	-7.528618341	Short	7.63
<i>Mergellus albellus</i>	Invertebrate	Non-hunted	5.7	-1.579129112	Short	6.41
<i>Mergus merganser</i>	Vertebrate	Non-hunted	7.3	-0.39697332	Short	7.28
<i>Mergus serrator</i>	Vertebrate	Non-hunted	7.3	-1.082901954	Short	6.92
<i>Bucephala clangula</i>	Invertebrate	Other hunted	8	-1.10518807	Short	6.82
<i>Melanitta nigra</i>	Invertebrate	Other hunted	7.5	2.279566759	Short	6.96
<i>Melanitta fusca</i>	Invertebrate	Other hunted	7.5	0.669883387	Short	7.50
<i>Tadorna tadorna</i>	Invertebrate	Non-hunted	10.9	4.30007485	Short	7.05
<i>Tadorna ferruginea</i>	Plant	Non-hunted	10.9	8.879664277	Res	7.12
<i>Netta rufina</i>	Plant	Hunted duck	7	5.352488457	Short	7.02
<i>Aythya fuligula</i>	Omnivore	Hunted duck	7.3	10.45232776	Short	6.55
<i>Aythya marila</i>	Invertebrate	Hunted duck	8.2	0.428197756	Short	6.91
<i>Aythya ferina</i>	Plant	Hunted duck	7.6	11.33614757	Short	6.71
<i>Aythya nyroca</i>	Plant	Non-hunted	7.6	9.773350072	Short	6.35
<i>Marmaronetta angustirostris</i>	Plant	Non-hunted	7	5.953747563	Short	6.17
<i>Mareca penelope</i>	Plant	Hunted duck	6.4	16.53006205	Short	6.65



<i>Mareca strepera</i>	Plant	Hunted duck	7.1	11.73496464	Res	6.82
<i>Anas acuta</i>	Plant	Hunted duck	6.8	16.89718621	Short	6.85
<i>Anas crecca</i>	Plant	Hunted duck	6.3	12.72483539	Short	5.84
<i>Anas platyrhynchos</i>	Omnivore	Hunted duck	6.6	-0.020462632	Short	6.74
<i>Spatula querquedula</i>	Omnivore	Hunted duck	6.5	23.06512822	Long	5.79
<i>Spatula clypeata</i>	Invertebrate	Hunted duck	6.5	14.91201976	Short	6.42
<i>Branta bernicla</i>	Plant	Non-hunted	10.9	2.863382582	Short	7.15
<i>Branta leucopsis</i>	Plant	Non-hunted	10.5	1.685984156	Short	7.43
<i>Anser albifrons</i>	Plant	Non-hunted	11.3	2.533119452	Short	7.83
<i>Anser fabalis</i>	Plant	Other hunted	11.4	-2.514040367	Short	7.92
<i>Anser brachyrhynchus</i>	Plant	Non-hunted	11.4	2.021202386	Short	7.88
<i>Anser anser</i>	Plant	Other hunted	11.6	4.465428485	Short	8.10
<i>Cygnus columbianus</i>	Plant	Non-hunted	12.7	2.436899298	Short	8.76
<i>Cygnus cygnus</i>	Plant	Non-hunted	12.3	-1.490706698	Short	9.14
<i>Cygnus olor</i>	Plant	Non-hunted	12.2	1.269147723	Res	9.28
<i>Platalea leucorodia</i>	Invertebrate	Non-hunted	7.2	17.95014398	Short	7.53
<i>Plegadis falcinellus</i>	Invertebrate	Non-hunted	6.7	23.13696657	Long	6.44
<i>Ciconia nigra</i>	Vertebrate	Non-hunted	15.9	22.98346583	Long	7.98
<i>Ciconia ciconia</i>	Vertebrate	Non-hunted	16.5	22.94906346	Long	8.15
<i>Gavia stellata</i>	Vertebrate	Non-hunted	9.8	3.054206132	Short	7.30
<i>Gavia arctica</i>	Vertebrate	Non-hunted	9.8	0.763783716	Short	7.72
<i>Gavia immer</i>	Vertebrate	Non-hunted	9.8	-0.043080125	Short	8.51
<i>Nycticorax nycticorax</i>	Vertebrate	Non-hunted	8.8	23.47202582	Long	6.70
<i>Ardeola ralloides</i>	Vertebrate	Non-hunted	6.7	23.78935548	Long	5.66
<i>Ardea cinerea</i>	Vertebrate	Non-hunted	10.3	18.73373742	Short	7.28
<i>Ardea purpurea</i>	Vertebrate	Non-hunted	10.5	23.86808726	Long	6.97

<i>Bubulcus ibis</i>	Invertebrate	Non-hunted	8.1	22.96031792	Res	5.91
<i>Casmerodius albus</i>	Vertebrate	Non-hunted	9.1	21.85591657	Short	6.77
<i>Egretta garzetta</i>	Invertebrate	Non-hunted	6.6	21.26252234	Long	5.75
<i>Ixobrychus minutus</i>	Invertebrate	Non-hunted	4.1	24.01168239	Long	4.61
<i>Botaurus stellaris</i>	Vertebrate	Non-hunted	5.5	18.39468171	Res	7.19
<i>Phalacrocorax aristotelis</i>	Vertebrate	Non-hunted	8.8	2.298892	Res	7.48
<i>Phalacrocorax pygmeus</i>	Vertebrate	Non-hunted	8.1	2.744268717	Short	6.52
<i>Phalacrocorax carbo</i>	Vertebrate	Non-hunted	11.3	18.40826409	Short	7.84
<i>Lymnocyptes minimus</i>	Invertebrate	Other hunted	5.4	20.02804147	Short	3.93
<i>Limosa limosa</i>	Invertebrate	Other hunted	8.6	21.01453888	Long	5.67
<i>Limosa lapponica</i>	Invertebrate	Non-hunted	8.9	19.22660394	Short	5.68
<i>Arenaria interpres</i>	Invertebrate	Non-hunted	7.3	17.86138751	Short	4.92
<i>Calidris alpina</i>	Invertebrate	Non-hunted	8.1	11.83058422	Short	3.97
<i>Calidris maritima</i>	Invertebrate	Non-hunted	7.6	-2.482943324	Short	4.19
<i>Calidris minuta</i>	Invertebrate	Non-hunted	6.8	22.77386732	Long	3.10
<i>Calidris alba</i>	Invertebrate	Non-hunted	8.1	18.86150911	Long	3.96
<i>Calidris temminckii</i>	Invertebrate	Non-hunted	7.1	23.31781746	Long	3.18
<i>Calidris ferruginea</i>	Invertebrate	Non-hunted	7.6	23.43021111	Long	4.08
<i>Calidris canutus</i>	Invertebrate	Other hunted	6.8	19.07810312	Long	4.96
<i>Tringa ochropus</i>	Invertebrate	Non-hunted	5.6	21.09896305	Short	4.28
<i>Tringa erythropus</i>	Invertebrate	Other hunted	5.6	21.03462626	Long	5.07
<i>Tringa nebularia</i>	Invertebrate	Other hunted	6.3	23.15678175	Long	5.24
<i>Tringa glareola</i>	Invertebrate	Non-hunted	5.2	23.5308322	Long	4.14
<i>Tringa totanus</i>	Invertebrate	Other hunted	6.2	15.48429434	Short	4.87
<i>Tringa stagnatilis</i>	Invertebrate	Non-hunted	5.6	23.22543672	Long	4.36
<i>Actitis hypoleucos</i>	Omnivore	Non-hunted	6.8	23.15678175	Long	3.89

<i>Scolopax rusticola</i>	Invertebrate	Other hunted	6.3	4.242742	Short	5.73
<i>Gallinago gallinago</i>	Invertebrate	Other hunted	4.8	18.58383523	Short	4.74
<i>Numenius arquata</i>	Omnivore	Other hunted	5	18.88781824	Short	6.69
<i>Numenius phaeopus</i>	Invertebrate	Other hunted	9.1	22.05833801	Long	5.90
<i>Larus minutus</i>	Invertebrate	Non-hunted	10.5	3.7	Short	4.78
<i>Larus genei</i>	Omnivore	Non-hunted	10.5	11.89835354	Short	5.64
<i>Larus ridibundus</i>	Invertebrate	Other hunted	9.6	6.581840595	Short	5.65
<i>Larus canus</i>	Omnivore	Non-hunted	9.8	1.617542777	Short	6.02
<i>Larus marinus</i>	Omnivore	Non-hunted	12	-2.491862384	Short	7.41
<i>Larus michahellis</i>	Vertebrate	Non-hunted	13	4.57847	Short	7.00
<i>Larus fuscus</i>	Vertebrate	Non-hunted	13.9	18.5800917	Short	6.64
<i>Larus audouinii</i>	Vertebrate	Non-hunted	13	11.45294748	Short	6.28
<i>Larus melanocephalus</i>	Vertebrate	Non-hunted	10.1	9.305989705	Short	5.55
<i>Rissa tridactyla</i>	Omnivore	Non-hunted	12.9	4.798985422	Res	6.03
<i>Sterna caspia</i>	Vertebrate	Non-hunted	12.2	21.40258537	Long	6.49
<i>Sterna hirundo</i>	Vertebrate	Non-hunted	11.5	25.24947703	Long	4.87
<i>Sterna sandvicensis</i>	Vertebrate	Non-hunted	10.8	12.38123263	Short	5.30
<i>Chlidonias hybrida</i>	Vertebrate	Non-hunted	9.9	21.65184602	Long	4.44
<i>Charadrius alexandrinus</i>	Invertebrate	Non-hunted	5	19.1981442	Short	3.77
<i>Vanellus vanellus</i>	Invertebrate	Other hunted	9	4.516681624	Short	5.39
<i>Charadrius dubius</i>	Invertebrate	Non-hunted	5	23.79976584	Long	3.68
<i>Charadrius hiaticula</i>	Invertebrate	Non-hunted	5.1	22.10771097	Short	4.18
<i>Recurvirostra avosetta</i>	Invertebrate	Non-hunted	8.7	22.04492438	Short	5.72
<i>Himantopus himantopus</i>	Invertebrate	Non-hunted	7.3	22.12582625	Long	5.18
<i>Haematopus ostralegus</i>	Invertebrate	Other hunted	13.7	15.34138378	Short	6.27
<i>Pluvialis squatarola</i>	Invertebrate	Other hunted	6	18.83630341	Short	5.53

<i>Pluvialis apricaria</i>	Invertebrate	Non-hunted	5.2	5.356544857	Short	5.37
<i>Fulica cristata</i>	Plant	Non-hunted	7	22.58950745	Res	6.72
<i>Porzana porzana</i>	Omnivore	Non-hunted	2.7	23.546673	Long	4.48
<i>Gallinula chloropus</i>	Omnivore	Other hunted	5.9	16.30709877	Res	5.83
<i>Fulica atra</i>	Plant	Other hunted	7	5.864663198	Res	6.58
<i>Porphyrio porphyrio</i>	Plant	Non-hunted	5.9	22.11989646	Res	6.65
<i>Rallus aquaticus</i>	Omnivore	Other hunted	4.6	4.523513575	Res	4.72
<i>Grus grus</i>	Plant	Non-hunted	13.4	14.71592338	Short	8.61
<i>Phoenicopterus roseus</i>	Omnivore	Non-hunted	16.3	20.58416977	Short	8.02
<i>Tachybaptus ruficollis</i>	Invertebrate	Non-hunted	5.4	18.5962031	Short	5.14
<i>Podiceps auritus</i>	Vertebrate	Non-hunted	7.1	3.393542644	Short	6.12
<i>Podiceps grisegena</i>	Omnivore	Non-hunted	7.1	2.273349328	Short	6.93
<i>Podiceps cristatus</i>	Vertebrate	Non-hunted	7.1	10.08300404	Short	6.60
<i>Podiceps nigricollis</i>	Invertebrate	Non-hunted	7.1	17.99795243	Short	6.02

700 Appendix 3. Sensitivity analysis.

701

702 1) Protected areas

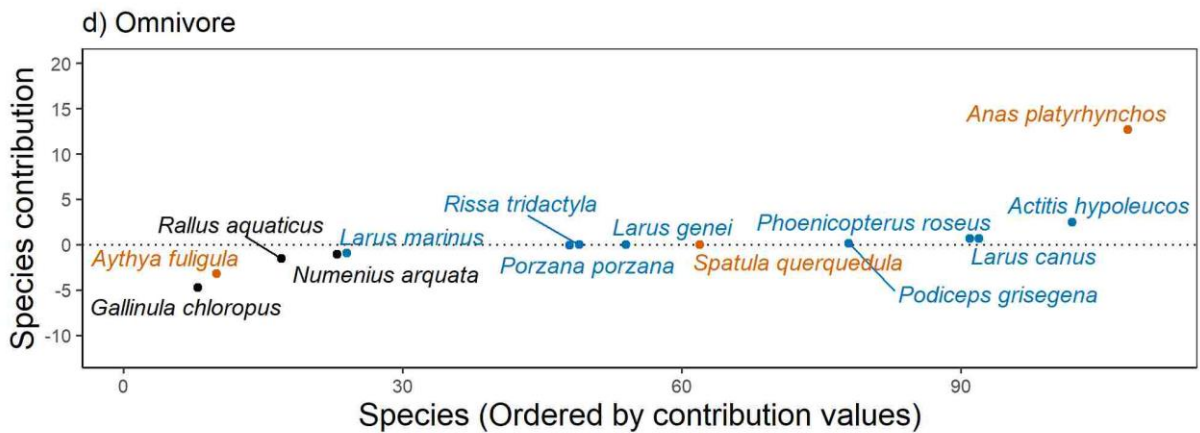
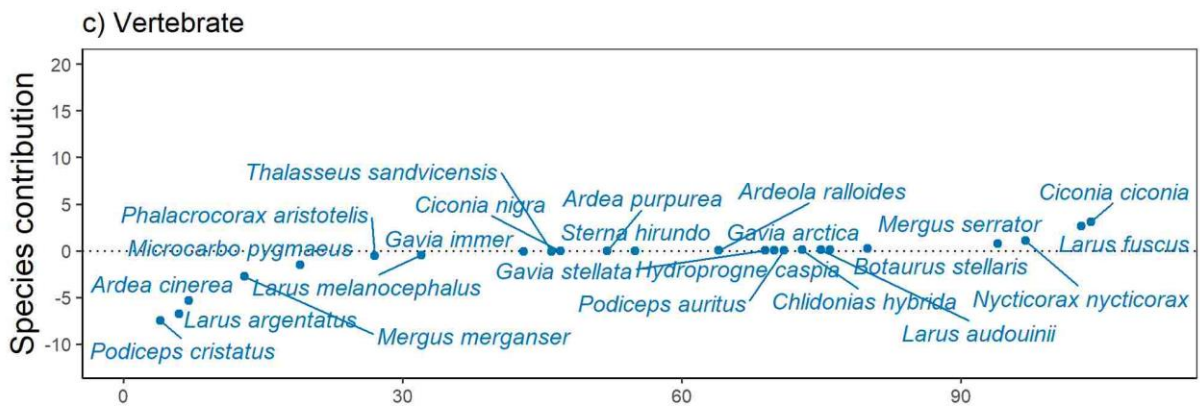
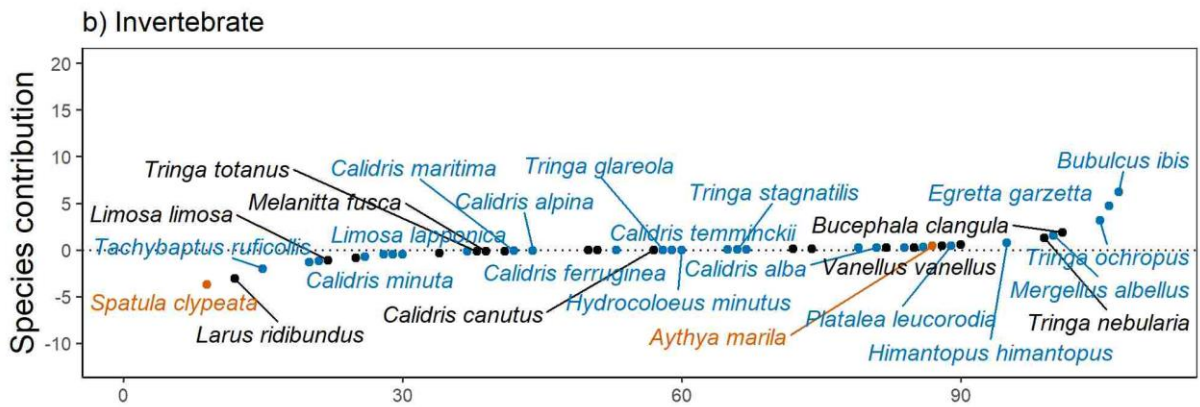
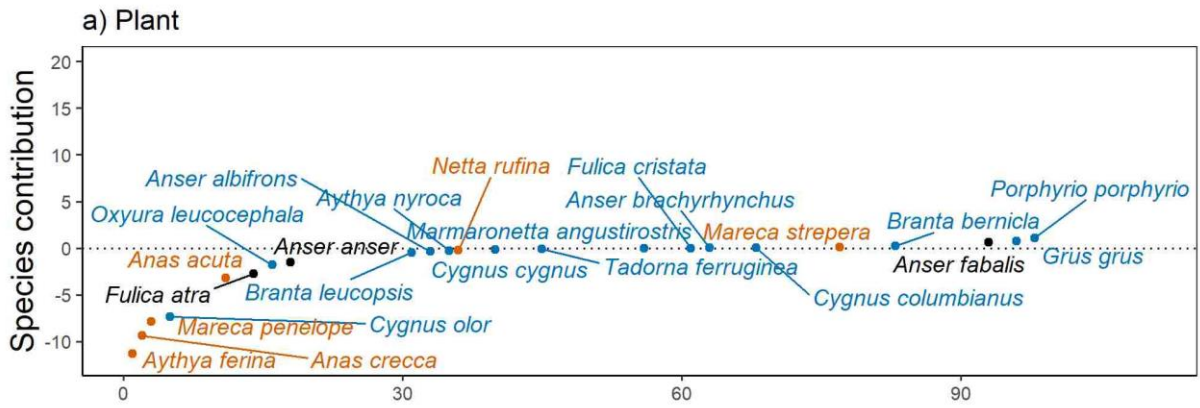
703

704 We performed a sensitivity analysis to ensure that the results from the main text were not affected by  
705 site protection status. Indeed, protected areas can affect wintering waterbird response to climate  
706 warming, by increasing species persistence (Gaget al. 2021). The risk is then to observe the effect of  
707 site protection on wintering waterbird community change rather than the potential effect of hunting  
708 management. It is important however to note that hunting activities can occur inside protected areas.  
709 This means that we cannot dissociate the effect of the protected area and the potential effect of  
710 hunting management inside protected areas. To ensure that our results were not affected by protected  
711 areas, we removed all protected sites from this sensitivity analysis. Protected sites were identified  
712 following the methodology described in Gaget et al. (2021). We performed the same analyses, without  
713 protected sites, as in the main text. We investigated temporal change in Community Temperature  
714 Index (CTI), species contributions and assessed the differences of contribution between species,  
715 depending on hunting status and functional traits.

716

717 We identified 411 monitored sites outside a protected area, totalling 8,229 surveys. The CTI temporal  
718 trend was positive and significant ( $\beta = 0.042 \pm 0.002$  SE,  $p < 0.001$ ). Species contributions to the CTI  
719 trends were fairly similar to the full analysis (Fig. A3.1). Species diet, hunting status and generation  
720 length were the main factors explaining species contribution, whereas other predictors had little effect  
721 (Table A3.1). Multicollinearity was low between species diet and hunting status (VIF = 1.09) and  
722 between species diet and generation length (VIF = 1.14). Consequently, this sensitivity analysis  
723 confirmed that species diet and hunting status are important factors explaining species contribution  
724 to CTI trend independently from site protection. In addition, generation length significantly  
725 contributed to explain species contributions in this subset of sites. Generation length can indeed be an  
726 important factor together with site fidelity, because juvenile settlement can influence over-winter  
727 location at adult stage, meaning that long-lived species might need more time to shift their distribution  
728 than short-lived species (see general discussion).

729



731 Figure Appendix 3.1: Species contributions (%) to the Community Temperature Index (CTI) temporal  
732 trend, depending on species diet (plant, invertebrate, vertebrate, omnivore) and hunting status  
733 (hunted ducks, other hunted and non-hunted), with species ordered by their contribution value (n =  
734 110 species). The horizontal dotted line represents a null contribution. Not all species names are shown  
735 for esthetic reason.

736  
737  
738  
739

740 Table Appendix 3.1. Model selection identifying predictors the most related to species contributions  
 741 based on AIC (delta AIC<2) and parsimony. The F-value corresponds to the sum of the effects and DF  
 742 is degrees of freedom. The best model to explain species contribution considers both species hunting  
 743 status, generation length and diet (in bold). Models are fitted with phylogenetic generalized least  
 744 squares (PGLS). Diet, hunting status and migration are factors and STI (Species Temperature Index),  
 745 body mass and generation length are quantitative variables.

746

Formula	AIC	Delta AIC	Pseudo R-squared	F-value	DF
<b>Hunting status + Diet + Generation length</b>	<b>699.7</b>	<b>0.0</b>	<b>0.105</b>	<b>7.574</b>	<b>7</b>
Hunting status + Diet + Body mass	700.5	0.8	0.083	4.901	7
Hunting status + Diet + Migration	701.1	1.4	0.062	2.512	8
Diet + Hunting status	701.8	2.1	0.062	2.514	6
Diet + Generation length	704.9	5.2	0.085	6.095	5
Diet + Migration	705.5	5.8	0.050	1.830	6
Diet + Body mass	705.7	6.0	0.063	3.389	5
Diet	706.2	6.5	0.048	1.794	4
Hunting status + Diet + STI	707.3	7.6	0.062	2.516	7
Hunting status	709.5	9.8	0.010	0.521	3
Diet + STI	711.7	12.0	0.049	1.836	5
Migration	712.4	12.7	0.005	0.294	3
Null model	713.5	13.8	0.000	0.000	1
Body mass	714.2	14.5	0.005	0.574	2
Generation length	714.4	14.7	0.019	2.041	2
STI	719.1	19.4	0.001	0.020	2

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750

## 751 2) Sites with more than 25 surveys over 28 years

752

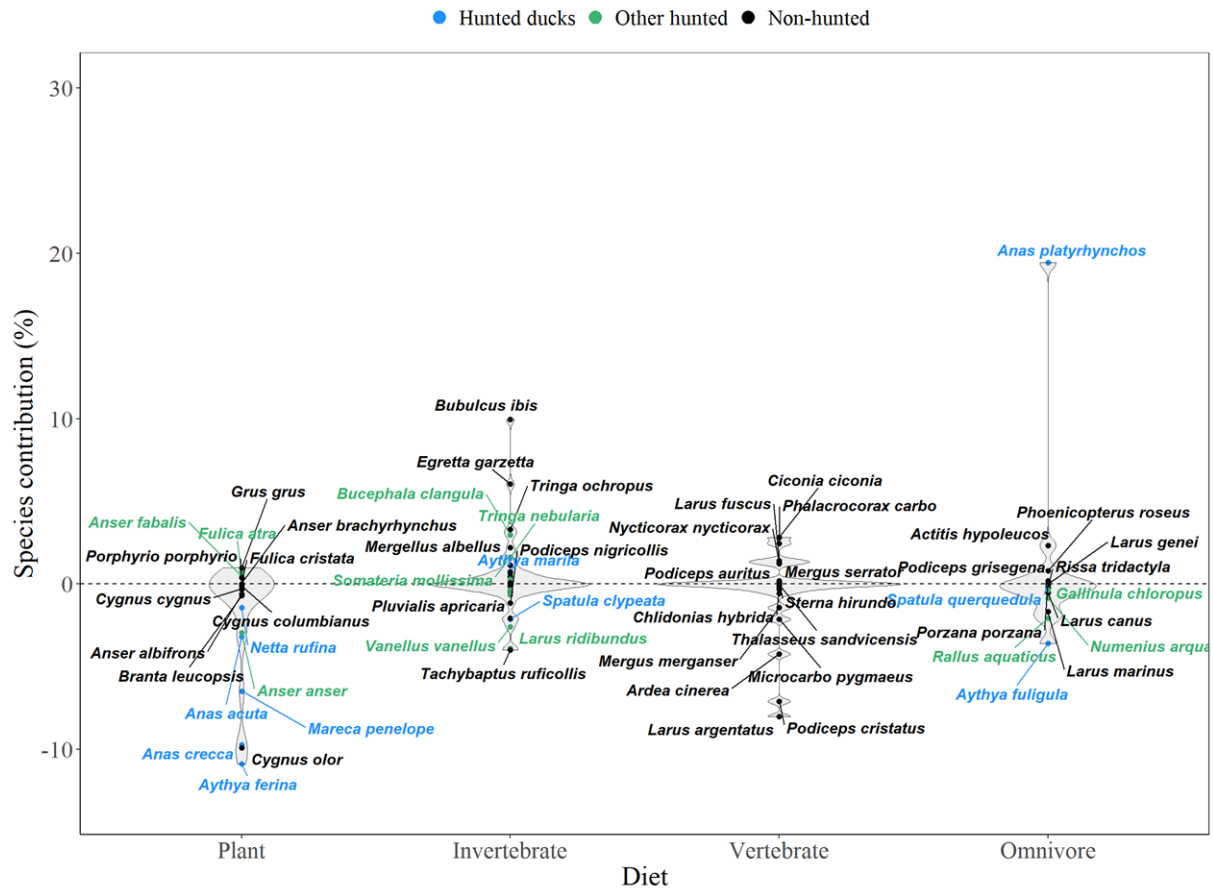
753 We performed a sensitivity analysis to ensure that the results from the main text were not affected by  
 754 the number of surveys per site. We removed all sites with less than 25 surveys from this sensitivity  
 755 analysis and performed the same analyses as in the main text. We investigated temporal change in  
 756 Community Temperature Index (CTI), species contributions and assessed the differences of  
 757 contribution between species, depending on hunting status and functional traits.

758

759 We identified 288 monitored sites outside a protected area, totalling 7,703 surveys. The CTI temporal  
 760 trend was positive and significant ( $\beta = 0.053 \pm 0.002$  SE,  $p < 0.001$ ). Species contributions to the CTI  
 761 trends were fairly similar to the full analysis (Fig. A3.2). Species diet, hunting status and generation  
 762 length were the main factors explaining species contribution, whereas other predictors had little effect  
 763 (Table A3.2). Multicollinearity was low between species diet and hunting status (VIF = 1.05).  
 764 Consequently, this sensitivity analysis confirmed that species diet and hunting status are important  
 765 factors explaining species contribution to CTI trend independently from site survey effort.

766





767  
 768 Figure Appendix 3.2 : Violin plot of species contributions (%) to the Community Temperature Index  
 769 (CTI) temporal trend, depending on species diet. Color scale indicates hunting status; hunted ducks  
 770 (blue), other hunted species (green), non-hunted species (black). Not all species names are shown for  
 771 esthetic reason.

772  
 773  
 774 Table Appendix 3.2. Model selection identifying predictors the most related to species contributions  
 775 based on AIC (delta AIC<2) and parsimony. The F-value corresponds to the sum of the effects and DF  
 776 is degrees of freedom. The best model to explain species contribution considers both species hunting  
 777 status, generation length and diet (in bold). Models are fitted with phylogenetic generalized least  
 778 squares (PGLS). Diet, hunting status and migration are factors and STI (Species Temperature Index),  
 779 body mass and generation length are quantitative variables.

780  
 781

Formula	AIC	Delta AIC	Pseudo R-squared	F-value	DF
<b>Hunting status + Diet + Migration</b>	<b>690.7</b>	<b>0.0</b>	<b>0.112</b>	<b>4.669</b>	<b>8</b>
Hunting status + Diet + Body mass	691.0	0.3	0.122	6.173	7
Diet + Hunting status	691.5	0.8	0.108	4.528	6
Hunting status + Diet + Generation length	692.1	1.4	0.129	7.001	7
Diet + Migration	695.5	4.8	0.094	3.654	6
Diet	696.2	5.5	0.092	3.567	4

Hunting status + Diet + STI	696.2	5.5	0.116	5.483	7
Diet + Body mass	696.4	5.7	0.100	4.541	5
Diet + Generation length	697.2	6.5	0.108	5.558	5
Diet + STI	701.1	10.4	0.098	4.326	5
Hunting status	704.4	13.7	0.010	0.543	3
Migration	707.9	17.2	0.001	0.022	3
Null model	708.3	17.6	0.000	0.000	1
Body mass	709.5	18.8	0.001	0.073	2
Generation length	711.0	20.3	0.003	0.365	2
STI	712.9	22.2	0.001	1.061	2

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784 Appendix 4. Model selection identifying predictors the most related to species contributions based  
 785 on AIC (delta AIC<2) and parsimony. The F-value corresponds to the sum of the effects and DF is degree  
 786 of freedom. The best model to explain species contribution considers both species hunting status and  
 787 diet (in bold). Models are fitted with phylogenetic generalized least squares (PGLS). Diet, hunting status  
 788 and migration are factors and STI (Species Temperature Index), body mass and generation length are  
 789 quantitative variables.

Formula	AIC	Delta AIC	Pseudo R-squared	F-value	DF
Hunting status + Diet + Generation length	695.8	0.0	0.097	6.835	7
Hunting status + Diet + Body mass	696.3	0.5	0.077	4.410	7
Hunting status + Diet + Migration	696.5	0.7	0.060	2.410	8
<b>Diet + Hunting status</b>	<b>697.2</b>	<b>1.4</b>	<b>0.058</b>	<b>2.329</b>	<b>6</b>
Diet + Generation length	700.3	4.5	0.083	5.879	5
Diet + Migration	700.6	4.8	0.049	1.849	6
Diet + Body mass	700.9	5.1	0.061	3.328	5
Diet	701.3	5.5	0.047	1.757	4
Hunting status + Diet + STI	702.9	7.1	0.058	2.323	7
Hunting status	704.8	9.0	0.007	0.36	3
Diet + STI	707.0	11.2	0.048	1.751	5
Migration	707.5	11.7	0.005	0.273	3
Null model	708.4	12.6	0.000	0.001	1
Body mass	709.2	13.4	0.004	0.455	2
Generation length	709.6	13.8	0.016	1.804	2
STI	714.0	18.2	0.001	0.052	2

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