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Abstract

There is a strong need for feedback in conservation policy. Recently, its importance has increased due to climate changes causing remarkable shifts in species distributions; such shifts could shape the effectiveness of a predefined protected area network. Based on twelve-year´s citizen-monitoring data (2004–2015) investigated by the legislative-based protected area network (in European Union called Special Protection Areas - SPAs), we evaluated the effectiveness of the network for 28 wintering waterbird species in a central European country, where total numbers are mostly increasing in recent decades. We test the hypothesis that SPAs protect wetland areas suitable for increasing wintering waterbird species. To this end, we use two different approaches: (i) long-term trend and species-specific variables explaining the proportions of numbers in SPAs at the multi-species level and (ii) individual-species changes in numbers inside and outside SPAs. The annual proportions of numbers recorded inside SPAs has been decreasing in studied species from 2004 to 2015 and has not increased as rapidly as the increase in numbers. Within eco-taxonomic groups, we show the high proportion of geese recorded inside SPAs, even though a higher rate of increase in numbers outside SPAs was found in some goose species (Great White-fronted Goose Anser albifrons and Greylag Goose Anser anser). Conversely, fish-eaters and diving ducks generally show a low preference for SPAs and yet fish-eating Great Cormorant Phalacrocorax carbo and Grey Heron Ardea cinerea show a higher increase in numbers inside SPAs. Feeding opportunities for expanding species (e.g. geese) in areas outside the protected network most likely exceed the advantages of reduced disturbance in SPAs; on the other hand, the reduced disturbance could be pivotal in possible conflict species (e.g. Great Cormorant and Grey Heron). An overall high proportions of numbers in SPAs for protected species was not confirmed; even EU criteria species do not show a significantly higher increase in numbers inside SPAs, except Smew. The hypothesis assuming SPAs as appropriate areas for wintering waterbirds was not confirmed. Based on two exploitable approaches, the study uses the data of a long-term, citizen, science-monitoring programme to indicate that a high proportions of numbers of individual species in a protected network should not necessarily mean positive changes in species numbers inside the network and vice versa. Recent climate-driven changes in species distributions very likely requires a flexible conservation policy, with decision-making and planning-strategies based on actual monitoring data, and full international cooperation. In light of this, we highlight the enormous importance of volunteer monitoring based on the annual efforts of nonprofessional ornithologists.

Keywords	conservation policy; changes in distribution; protected areas; volunteer monitoring; wetlands; wintering numbers
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Dear Editor,
We send you the manuscript entitled:
“Importance of Special Protection Areas for wintering waterbirds evaluated by annual citizen science-monitoring programme: multi-species and individual-species approach”

Based on twelve-year´s of volunteers´data, we evaluate using two approaches the effectiveness of Special Protection Areas for wintering waterbird assemblages in a central European country, where total numbers are increasing in recent decades. We highlight the necessity of extension of the network; hence the increase in numbers could be more rapid than proportion of numbers in SPAs.

We decided to submit the manuscript considering the feedback of conservation policy to Biological Conservation in line of the journals ‘aims and scopes emphasized.

We would like to state that (i) the manuscript has not been published or submitted for publication elsewhere and (ii) all authors have contributed to designing and/or performing the research and writing the manuscript, and have read and approved the manuscript prior to submission.

Yours sincerely,

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Importance of Special Protection Areas for wintering waterbirds evaluated by annual citizen science-monitoring programme: multi-species and individual-species approach

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Abstract

There is a strong need for feedback in conservation policy. Recently, its importance has increased due to climate changes causing remarkable shifts in species distributions; such shifts could shape the effectiveness of a predefined protected area network. Based on twelve-year´s citizen-monitoring data (2004–2015) investigated by the legislative-based protected area network (in European Union called Special Protection Areas - SPAs), we evaluated the effectiveness of the network for 28 wintering waterbird species in a central European country, where total numbers are mostly increasing in recent decades. We test the hypothesis that SPAs protect wetland areas suitable for increasing wintering waterbird species. To this end, we use two different approaches: (i) long-term trend and species-specific variables explaining the proportions of numbers in SPAs at the multi-species level and (ii) individual-species changes in numbers inside and outside SPAs. The annual proportions of numbers recorded inside SPAs has been decreasing in studied species from 2004 to 2015 and has not increased as rapidly as the increase in numbers. Within eco-taxonomic groups, we show the high proportion of geese recorded inside SPAs, even though a higher rate of increase in numbers outside SPAs was found in some goose species (Great White-fronted Goose *Anser albifrons* and Greylag Goose *Anser anser*). Conversely, fish-eaters and diving ducks generally show a low preference for SPAs and yet fish-eating Great Cormorant *Phalacrocorax carbo* and Grey Heron *Ardea cinerea* show a higher increase in numbers inside SPAs. Feeding opportunities for expanding species (e.g. geese) in areas outside the protected network most likely exceed the advantages of reduced disturbance in SPAs; on the other hand, the reduced disturbance could be pivotal in possible conflict species (e.g. Great Cormorant and Grey Heron). An overall high proportions of numbers in SPAs for protected species was not confirmed; even EU criteria species do not show a significantly higher increase in numbers inside SPAs, except Smew. The hypothesis assuming SPAs as appropriate areas for wintering waterbirds was not confirmed. Based on two exploitable approaches, the study uses the data of a long-term, citizen, science-monitoring programme to indicate that a high proportions of numbers of individual species in a protected network should not necessarily mean positive changes in species numbers inside the network and vice versa. Recent climate-driven changes in species distributions very likely requires a flexible conservation policy, with decision-making and planning-strategies based on actual monitoring data, and full international cooperation. In light of this, we highlight the enormous importance of volunteer monitoring based on the annual efforts of nonprofessional ornithologists.

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Introduction

Migratory birds require effective management for critical sites throughout their annual cycle (Pullin 2002, Hegemeijer 2006, Donald et al. 2007, Kirby et al. 2008) including the conservation of non-breeding areas (Sutherland et al. 2004). Sustainable conservation, in particular migratory bird conservation, still presents unresolved problems that call for decisions based on scientific research. Such research - often based on volunteers´ monitoring efforts - should also make it possible to assess the effectiveness of conservation measures taken internationally (Sutherland et al. 2004), as recognised in the text of the globally-respected Convention on Biological Diversity (www.cbd.int). International conservation policy at the flyway level can bring measurable conservation benefits for species (Pullin 2002, Sinclair et al. 2006, Donald et al. 2007), not least because international cooperation is essential throughout the areas used by populations (O´Connell et al. 2006, Hagemeijer 2006). As a consequence of this 'flyway approach' (for term definition, see Boere and Stroud 2006, Hagemeijer 2006; see also Lehikoinen et al. 2013, Pavón-Jordán et al. 2015), the European Union's legislation ensures biodiversity conservation through the Birds and Habitat Directives (The Council Directive 2009/147/EC). The Birds Directive requires Member States to select the most suitable sites and designate them as Special Protection Areas (SPAs). Sufficient sites need to be designated so as to form a coherent network for vulnerable and migratory species throughout the annual cycle (Donald et al. 2007). This SPA network is also being used to protect wintering populations and their environment; however it was not primarily designed for the purposes of wintering waterbirds.

The current networks of worldwide protected areas (Chape et al. 2008) could soon become inadequate in light of recent climate changes and the corresponding distributional shifts in wintering ranges (Lovejoy 2006, Thomas et al. 2012, Guillemain et al. 2013, Mason et al. 2015). Waterbird species are already responding to rapid changes in climate (Crick 2004, Møller et al. 2010) as seen by altered distributions and numbers (Thomas and Lennon 1999, Maclean et al. 2008, Lehikoinen et al. 2013). Even given the strong necessity for adequate feedback from conservation policy (Sutherland et al. 2004, Donald et al. 2007, Møller et al. 2010, Albuquerque et al. 2013, Lehikoinen et al. 2013, Pavón-Jordán et al. 2015, Thomas and Gillingham 2015), relatively few studies are evaluating the effectiveness of the protected area network in its non-breeding areas in the light of range changes and the increasing effect of climate change on birds (Johnston et al. 2013, Pavón-Jordán et al. 2015).

Here, we focus on wintering waterbirds as an internationally important bird assemblage (Gilissen et al. 2002), a group that has been pivotal to the legal classification of SPAs in Europe (Heath et al. 2000). For evaluating the effectiveness of this SPA network, we have looked into the twelve-year period immediately since the Birds Directive (The Council Directive 2009/147/EC) was implemented in the Czech legislation, as an example of a Central European Member State integration since 2004. After the Directive's implementation, 41 selected areas were expected to ensure species protection in the Czech Republic, including wintering grounds (Chvátal 2009). Even though the majority of wintering grounds have traditionally been found in the coastal areas of northwest Europe, the Baltic Sea and the Mediterranean region (Gilissen et al. 2002, Rendón et al. 2008, Jackson et al. 2009, van Roomen et al. 2012), the importance of central Europe has been increasingly recognized for wintering waterbird populations in recent decades (Fox et al. 2010, Keller 2011, Musil et al. 2011, Pavón-Jordán et al. 2015); a change likely attributable to distributional shifts caused by recent climate change (Lehikoinen et al. 2013). We aim to contribute to some key questions: which species prefer the existing SPA network and how effective has it been for individual species over the twelve-year period when considering the modified distribution attributed to climate change? We used the monitoring data of wintering waterbirds - the International Waterbirds Census (later IWC) during the twelve-year period since implementation of the

Birds Directive in the study area (2004–2015). This monitoring programme, though volunteer-based, has a long tradition and is regularly organised; it is mainly aimed at population size estimates and individual wetlands importance assessment (Gilissen et al. 2002, Wetlands International 2015).

Based on detailed records of individual wetlands in the Czech Republic, we use two approaches as an aid to establish the importance of wetlands inside the protected area network (SPAs), both at the multi-species and individual-species level. In the first step, species-specific variables (i.e. conservation status, population size and trend, geographical distribution, water-type specialisation and eco-taxonomic group) were supposed to answer the question: which species prefer the protected site network by assessing the annual proportions of numbers in SPAs. In the second step, individual-species trends in numbers calculated on wetlands inside and outside SPAs should indicate how particular species changed its distribution considering SPAs.

Protected areas should act as special site refuges that facilitate both species' wintering requirements (Ridgill and Fox 1990, Pullin 2002, Sinclair et al. 2006) and the range expansions caused by recent climate change (Thomas et al. 2012). Therefore, we hypothesize that SPAs are currently protecting the appropriate areas (The Council Directive 2009/147/EC; see also Sutherland et al. 2004, Devictor et al. 2007, Donald et al. 2007, Thomas et al. 2012, Hiley et al. 2013, Smart et al. 2014, Kukkala et al. 2016). Based on this hypothesis, we draw three predictions. (1) We predict the long-term increase in proportions of numbers in SPAs relative to non-protected sites. (2) When species increasing in numbers, we predict higher increase in proportions of numbers in SPAs than outside SPAs. (3) In the individual-species level, we predict prevailing higher rate of increase or else lower rate of decrease in SPAs than outside SPAs.

Methods

Waterbird data

Count numbers for 28 of the most common waterbird species annually exceeding 50 wintering individuals in the study area (see Table 1 for list of species) were taken from results of the International Waterbird Census (IWC). IWC is a worldwide-coordinated census conducted by individual countries and organised by Wetlands International in mid-January each winter on predetermined dates and sites with the aim to maximize synchrony (Gillisen et al. 2002). The count date is considered the coldest period of winter when food and thermoregulatory effects on wintering species distribution is most apparent (Ridgill and Fox 1990, Dalby et al. 2013). About 350 volunteer birdwatchers annually contribute to the monitoring in our (Czech) study area. They are mostly non-professional ornithologists or voluntary professionals monitoring in their own time. The methodology requires a single count at each site each winter, optimally conducted by the same person in consequent winters. The high quality of the IWC data has been proved in recently published studies (e.g. Fox et al. 2010, Lehikoinen et al. 2013, Musilová et al. 2014, Pavón-Jordán et al. 2015, Musilová et al. 2015). We analysed the records of 991 wetlands in the Czech Republic since the year 2004, when the Special Protection Areas were declared by the Czech government directives, up to the recent year 2015. The protected network covers 41 SPAs and 8.9 % of the total area of the Czech Republic (Chvátal 2009). In total, we included 120 wetlands located in SPAs and 871 wetlands outside the SPA network (Fig. 1). Counted wetlands were chosen in aim to achieve almost evenly coverage of the study area by monitoring scheme (Musilová et al. 2014), see Figure 1 for details. The wetlands ranged from both standing waters (reservoirs, fishponds, gravel and sand-pit lakes, and industrial settling ponds) and running waters (rivers and streams). For running waters, sites were defined as river sections with well-defined boundaries, such as dams, weirs and bridges (for the list of wetland habitats in Czech

Republic, see Chytil et al. 1999). The three gull species Herring Gull *Larus argentatus*, Caspian Gull *Larus cacchinnans* and Yellow-legged Gull *Larus michahellis* were termed 'large gulls',and hereafter are treated as one single species, in accordance with the former taxonomic situation valid at the beginning of the monitoring programme and regarding the possible problem with field identification (Rose 1995, Musil et al. 2011, Musilová et al. 2014, Wetlands International 2015,). Bird names follow the Avibase Clements Checklist (http://avibase.bsc-eoc.org).

Species-specific variables

All investigated waterbird species were described using seven species-specific variables that might explain their pattern of annual proportions of numbers in SPAs (see Table 1); these variables are defined below. (1) Species were divided into three groups according to their conservation and hunted status: *protected non-hunted species, hunted species and non-hunted species*. The conservation status of particular species was classified according to their listing in Annex I of the Birds Directive (Council Directive 2009/147/EC on the conservation of wild birds) as well as the classification of species under the Czech legislation Act of Protection of Nature and Landscape No. 114/92 Coll. and Regulation No. 395/1992 Coll., Annex No. III (list of Specially Protected Animals; Hudec et al. 1999). 'Hunted species' indicated a species allowed to be hunted in the Czech Republic (listed in Hunting Act No. 4 49/2001 Coll.). The waterbird hunting period finish before the census term in the study area as well as in neighbouring countries (Mooij 2010). (2) *The flyway population size* and (3) *Flyway population trends* (i.e. trend in numbers of a species in the flyway of the Western Palearctic) were obtained from Waterbird Population Estimates (Wetlands International 2015). For population trends in the Western Palearctic −1, 0 and +1 values were included, where −1 indicated a decreasing trend, 0 a stable population, and +1 an increasing trend. Moreover, estimates of breeding population size and trends in the breeding population (Birdlife International 2004) were used for White-tailed Sea-eagle *Haliaeetus albicilla*, Common Kingfisher *Alcedo atthis* and White-throated Dipper *Cinclus cinclus*, whose data are not included in Waterbird Population Estimates (Wetlands International 2015). These three species are both breeding and wintering in Europe (Snow and Perrins 1998) and therefore their total population size and population trends were taken from their breeding population data (Birdlife International 2004). (4) Time totals (see below) in an individual year were used as an estimate of numbers of wintering birds a species in the Czech Republic (henceforth *Czech population estimate*). (5) The geographical distribution of a species was classified using the *latitudinal midpoint* (Lemoine et al. 2007), i.e. the mean of the southernmost and northernmost latitudes of a species' breeding range (Snow and Perrins 1998). Latitudinal midpoint was used to explain the proportions of numbers in SPAs for species with different geographical range. (6) We consider the suitability of the SPA network for species inhabiting different wetland types. For each species, we therefore calculated an index of water-type specialization in the following manner. In accordance with Musil et al. (2011), we classified all sites into four habitat categories: rivers and streams, reservoirs, fishponds, and industrial waters. Next, we calculated the proportion of sites in each category (water type proportions) and the fraction of the given species' numbers that have been observed on sites of individual categories (count proportions). The *water type specialization index* is defined as Pearson's χ^2 statistic of the test of equivalence of (empiric) count proportions and (expected) habitat proportions. (7) Waterbird species were divided into six *eco-taxonomic groups*: fish-eating birds, geese, dabbling ducks, diving ducks, gulls, and others (see Snow and Perrins 1998), as used previously in Musil et al. (2011).

Statistical analysis

We used log-linear Poisson regression analysis to impute any missing 2011–2015 IWC waterbird count data from the long-term IWC data series (1966–2014) using Trends and Indices for Monitoring data (TRIM) software (Statistic Netherlands version 3.52, Pannekoek and Van Strien 2005) in two cases: to calculate the Czech population estimate (see also Musilova et al. 2014) and the individual-species trend in numbers. Regression parameters were estimated using generalized estimating equations (GEE). Missing data was usually the result of incomplete coverage due to limited availability of volunteers in some seasons. Serial correlations between annual numbers and over-dispersion in the data were also taken into account. The models used included change points to allow for changes in the slope parameters at some points in the time series (Pannekoek and Van Strien 2005, Fouque et al. 2007, 2009). 'Time Totals' values (hereafter used as Time Totals) of the IWC data (*i.e.* the actual count values plus the numbers of birds estimated for non-covered sites by the TRIM software) for all 991 sites included in the analysis were used to generate an estimate of the Czech population size of a species (termed Czech population estimate). The overall slope (i.e. the change in indices from one year to the next) was used to estimate the individual-species trend in numbers inside and outside the SPA network and then categorised depending on whether the rate of change was more or less than 5% per year: strong increase or decrease (>5% per year); a moderate increase or decrease (<5% per year); and a stable (trend is not significant and CIs were sufficiently narrow) or an uncertain trend (wide CI), see also Fouque et al. (2009) and Musil *et al.* (2011). The Wald test was used to test the significance of differences in rate of changes in numbers inside and outside the SPA network. We classified all investigated wetlands as SPA/non-SPA and use this category as an individual covariate in the linear trend models (see also Pavón-Jordán et al. 2015).

The number of individuals observed on SPA sites was modelled as a binomial outcome (termed *proportions of numbers in SPAs*), with the number of trials corresponding to the number of observed individuals of a given species in a given year. The effect of all the investigated variables was estimated using a multilevel (or mixed) generalized linear model with a logit link function and species-specific random effects. More concretely, we estimated the model

$$\mathrm{logit}(p_{is}) = \mathbf{x}_{is}\boldsymbol{\beta} + \varepsilon_s,$$

where p_{is} the probability that an individual i of species s is recorded on an SPA site (as opposed to a non-SPA site), $\mathbf{x}_{is}$ is the (row) vector of values of independent variables in an individual i of species s,[1] $\boldsymbol{\beta}$ is the estimated (column) vector of parameters, and ε_s is the species-specific random error. Due the nonlinear nature of the model, the values of β coefficients do not interpret easily. In order to facilitate the interpretation, we report the so-called average partial effects (*APE*) instead of the regression coefficients (Wooldridge, 2010). Such a measure is being used heavily in social sciences (Cameron and Trivedi 2005, Wooldridge 2010). For instance, if the *APE* on variable *A* is 15.5, a unit increase in *A* (with all other variables being held constant) is expected to increase the proportions of numbers in SPA by 15.5 percentage points.

As the number of independent variables is relatively high, we calculated variance inflation factors (VIFs) to see whether excessive inter-correlation might derail simultaneous use of all these variables in a multiple regression. The highest VIF value was 4.5 for the variable indicating protected non-hunted species; this is below the usual threshold value of 10, but still indicates some degree of correlation. Therefore, although we did use all independent variables in the regressions, we let them enter our regressions in a hierarchical fashion so as to be able to assess the stability of estimated coefficients (all hierarchical steps are reported in the Results section). Multilevel regressions were estimated in Stata 13 (StataCorp, College Station, TX).

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[1] The values of categorical variables (e.g., huntable species or group) were coded into a set of indicator variables for the sake of the regression.

Results

Species-specific preference of SPAs

Among the 120 investigated wetlands inside SPAs and 871 wetlands outside SPAs, we analysed the proportions of numbers of 28 species in SPAs using multilevel generalized linear models. Covering the twelve years since the Birds Directive implementation (2004–2015), the proportions of numbers in the SPA network was generally decreasing by 0.151% per year (Table 2, average partial effects on Year; $APE = -0.151$, $P < 0.001$). The decrease of SPA proportions is also noticeable from the trend curves in Figure 2 (e.g., in Common Teal *Anas crecca* and Eurasian Wigeon *Anas penelope*). Flyway population size (log-transformed) significantly affected a species' proportion in SPAs ($APE = -0.391$, $P < 0.001$). Species with a lower flyway population size showed a significantly higher proportion in SPAs by 0.391% (e.g. White-tailed Sea-eagle, Smew *Mergellus albellus* and Gadwall *Anas strepera* – Fig. 2). The proportions of numbers inside SPAs was higher in species with a higher Czech population estimate ($APE = 0.215$, $P < 0.001$). If the Czech population estimate of a species increases in numbers by 1%, the proportions of numbers in SPAs of the species increases only by 0.215%. Eco-taxonomic groups varied significantly with respect to proportions of numbers in SPAs (see Fig. 2 for details). Geese winter more frequently inside SPAs ($APE = 77.89$, $P < 0.05$); conversely, diving ducks ($APE = -0.45$, $P < 0.05$), others ($APE = -0.45$, $P < 0.05$; Mute Swan *Cygnus olor*, Little Grebe *Tachybaptus ruficollis*, Euarasian Moorhen *Galinulla chloropus*, Eurasian Coot *Fulica atra* and White-throated Dipper) and fish-eaters ($APE = -0.67$, $P < 0.05$) winter more frequently outside the SPA network. The protection/hunting status of the species, flyway population trend in numbers and water-type specialisation index showed no significant effect on proportions of numbers in SPAs (Table 2).

Individual-species efficacy of SPAs

Significant differences in trends in numbers inside and outside the SPA network were found in 15 of the 28 investigated species (see Table 3 for details); the trend inside SPAs showed a positive change in eight of them (predominantly fish-eating species) and a negative change in seven of them (predominantly herbivorous species). The overall fluctuating Smew, listed as an Annex I species, showed an increase inside while at the same showing a decrease outside SPAs. The Tufted Duck *Aythya fuligula* and Eurasian Coot were found stable in total and increasing inside SPAs. Higher increases in numbers inside SPAs than in their totals were found in Common Goldeneye *Bucephala clangula*, Goosander *Mergus merganser* and Great Crested Grebe *Podiceps cristatus*. The overall decreasing Great Cormorant *Phalacrocorax carbo* increased inside while decreasing outside SPAs, whereas the decreasing Grey Heron *Ardea cinerea* was stable inside and decreasing outside SPAs. Four herbivorous species (Great White-fronted Goose, Greylag Goose, Eurasian Wigeon *Anas penelope* and Gadwall) showed an increase in numbers in total while having significantly lower rate of increase inside SPAs. The overall stable Common Teal *Anas crecca* showed a decrease inside SPAs and an increase outside SPAs. More negative trends in numbers were found in the increasing Mew Gull *Larus canus* and the stable 'large Gulls', even though fluctuating inside SPAs. Among the three remaining Annex I species, Great White Egret *Ardea alba*, White-tailed Sea-eagle and Common Kingfisher showed no significant changes in numbers inside and outside SPAs.

Discussion

Evaluating the effectiveness of conservation policy and management among protected area networks is of considerable worldwide importance and should be undertaken using relevant sources of knowledge that cover the changes in distribution and numbers of species (Pullin 2002, Donald et al. 2007). In our study, we used the data of a citizen, science-monitoring programme of long tradition; this volunteer-based programme annually collects valuable information about the wintering distribution of waterbirds covering the whole study area. Based on two approaches, we test the hypothesis that the protected area network (SPAs) protects appropriate areas for the most common waterbird species. The first multi-species analysis helped to test the predictions that proportions of numbers in SPSs will increase over 12 year period and potentially increasing waterbird numbers will show higher increase in proportions of numbers inside than outside the SPA network. Actually, the prevailing increase in numbers was demonstrated at the individual-species level in the study area. The predictions was not supported, since the proportions of numbers slightly decrease over 12 years and the increase in numbers was found to be more rapid than the increase of proportions inside SPAs: a 1% increase in numbers compared to only a 0.21% increase of proportion inside SPAs. The second analysis brought more detailed insights to the first analysis, evaluating future changes in numbers of individual species. However, the predicted prevalence of higher rate of increase or lower rate of decrease inside SPAs was not showed, instead the positive and negative trends inside SPAs were almost equivalent. Given a defined annually-monitored study area, these two approaches are applicable for an assessment of the efficacy of protected areas for defined groups of species.

In our study area, the hypothesis that SPAs protect appropriate areas for wintering waterbirds was not confirmed using two approaches. The documentation for the designation of the Czech SPA network was prepared in 2002 with the aim of proposing the most important areas for breeding bird species, as well as areas with occurrences of migrating species - overwintering grounds, migration stopovers, gathering and moulting grounds (Chvátal 2009). Thus wintering waterbird assemblages were not disregarded in the conservation policy declaration for this given region. However, more recently, conservation policy has undoubtedly come under pressure due to the changes in species' ranges caused by climate change (Austin and

Rehfisch 2005, Lehikoinen et al. 2013, Brambilla et al. 2015, Pavón-Jordán et al. 2015); such climate change is expected to bring higher global mean temperatures and greater frequencies of extreme events (Lovejoy 2006, Beniston et al. 2007, IPCC 2007, Coumou and Rahmstorf 2012). No doubt the distributional shifts of waterbirds driven by climate change are likely to occur both more strongly and rapidly during the wintering period (Guillemain et al. 2013), while the distribution will become more temperature-dependent (Ridgill and Fox 1990, Adam et al. 2015). Here we have demonstrated that the rapidity of such waterbird distributional changes could therefore shape the effectiveness of conservation management when preferences for SPAs have not increased as much as species numbers (see also Rodrigues et al. 2004, Guillemain et al. 2013, Pavón-Jordán et al. 2015), particularly comparing wintering and breeding monitoring data. With regard to breeding populations, the important role of protected areas as establishment centres for range-shifting, newly-colonizing species has been shown for the UK (Hiley et al. 2013).

However, overall decreasing proportions of numbers inside SPAs, and decreasing numbers of some waterbird species in SPAs, does not necessarily mean that there has been a decrease in the quality of SPAs; rather it could simply be due to increases in the suitability of other sites outside the SPA network (Stillman et al. 2010). For the given region of the Czech Republic, in central Europe, the decreasing proportions of numbers in SPAs do not necessarily indicate some uncertainty in SPA designation. While numbers of waterbird species are mainly increasing here (Musil et al. 2011), this is a likely a consequence of the increasing importance of central Europe for wintering waterbirds (Fox et al. 2010, Keller and Burkhardt 2011, Pavón-Jórdan et al. 2015). At the same time, we note that in a previous study (Musilová et al. 2015) density-dependent regulation has been indicated, mean total numbers per site having not increased since the 1990s whereas waterbird numbers have been increasing in areas traditionally deemed ‘cold’. In line with these findings, we suppose that the further designation of some additional important sites (outside the current SPA network) that would cover species habitat requirements could help to protect waterbirds on their wintering grounds and increase the overall effectiveness of conservation management. Moreover, SPA designation should consider the behaviour and habitat requirements of the species concerned, as well as the effects of human disturbance (López-López et al. 2007, Briggs et al. 2012); certainly the population dynamics of species along with habitat changes bring together a more complicated issue (Stillman et al. 2010, Hiley et al. 2013, Guillemain et al. 2013).

Focusing on our four criteria species (Annex I), we demonstrated that the current SPA network does not generally serve as 'safe refuges' that facilitate their wintering requirements and their environmental- and climate-dependent range changes (Donald et al. 2007, Thomas et al. 2012). High proportions of numbers in SPAs were found in Smew, White-tailed Sea-eagle and Great White Egret. However, the long-term changes inside SPAs did not show significantly higher rates of increase in numbers when compared to sites outside the network for the Great White Egret, White-tailed Sea-eagle and Common Kingfisher. These species likely do not follow the advantages of SPAs covering reduced disturbance and human development pressure. The only exception was Smew, since trends in wintering numbers are positive in SPAs but negative outside SPAs. This finding is in line with the study of Pavón-Jordán et al. (2015) covering the north-eastern and south-western parts of the Smew flyway. Nevertheless, the overall legislative protection status of the species proved of low importance in SPA preference. However, by way of contrast, species of smaller populations would appear to prefer SPAs. Two of these low-population species (White-tailed Sea-eagle and Smew) belong to the highlighted species of Annex I.

Focusing in detail on geese and their wintering distributions, there is a significantly high preference, as well as high proportions of numbers, for SPAs. But, conversely, there has been a significantly higher increase in numbers outside SPAs, as shown for Great White-fronted and Greylag Goose. Wintering geese could therefore be moving to other suitable sites outside the SPAs due to the effects of density-dependent regulation within SPAs, as numbers of geese have been increasing both in the Western Palearctic and especially in central Europe (Madsen et al. 1999, van Eerden et al. 2005, Delany and Scott 2006, Fox et al. 2010, Musil et al. 2011) in recent decades. New unprotected wintering areas with sufficient feeding opportunities will most likely be of increasing relevance, especially when the requirement for sufficient ice-free freshwater could also be available (as highlighted in Adam et al. 2015, Musilová et al. 2015). Annually, wintering flocks of geese could comprise thousands of individuals (Musilová et al. 2014) since this may also explain the higher preference for SPAs – with the higher population size in the given area. As a consequence, increases in geese numbers outside SPAs could bring increasing damage to winter-crops in agriculture areas, and thus fuel conflicts with agro-economic interests (Jensen et al. 2008). Since no general, flexible framework for providing compensation in these areas currently exists, farmers′ efforts to control geese numbers feeding on crops could escalate. However, when we take into consideration the feeding biology of these species, geese form a unique group that feed outside wetland areas in winter and are thus not strictly dependent on the food composition of wetlands (Reed 1976, Fox et al. 2005, Gauthier et al. 2005). Geese fly to feeding grounds near first light and stay for most of the day (Owen and Black 1990). Moreover, waterbirds including geese are undoubtedly more greatly affected by disturbance in their wintering grounds (see review by Vickery and Gill 1999, Evans and Day 2001) and this fact could be the cause of the high preference of geese for SPAs serving as night roosts in our study area. Food accessibility seems to be an important factor affecting the distribution of wintering waterbirds (Newton 1998, Newton 2013) as has been previously indicated for dabbling ducks (Dalby et al. 2013). Most likely the accessibility of food in wetlands would explain the low preference of diving ducks for SPAs, as these groups feed strictly inside wetlands. Previous studies have confirmed a higher rate of increase in numbers in running rather than standing waters (Musil et al. 2011). Regardless of this, a more positive trend in numbers within SPAs rather than outside the network was found in Tufted Duck and Common Goldeneye among diving duck species. Similarly, for fish-eaters, their wetland-dependent food ecology and the low disturbance in SPAs could also shape their distribution, since this group exhibited a low SPA preference and yet, the reverse, a more positive trend in numbers within SPAs than outside. Given the differences in preferences and trends in numbers inside and outside SPAs (see above), our results likely support the previously-published findings of Dalby et al. (2013): food resources seem to be the main force shaping winter-site choices.

The European Unions´ Special Protection Areas (SPA) network represents the basis of habitat conservation for safeguarding populations of migratory waterbirds using East Atlantic flyway (Directive 2009/147/EC). Nevertheless, migratory waterbirds do not acknowledge state and site borders; the increasing waterbird numbers in our study area would seem to be the consequence of increasing numbers at the flyway level (Musil et al. 2011). However, this study indicates that the increases in numbers could be more rapid than increases in preference for SPAs and the proportions of numbers in SPAs are slightly decreasing over the study period. Hence, international cooperation in safeguarding areas is definitely relevant at the flyway level, since international coordination is required at the level of research, planning and monitoring, in common standards for legislation, protected area designation and management, and in the sharing of information (Hagemeijer 2006, Lehikoinen and Virkkala 2016). Studies that indicate the SPA network as not matching species distribution patterns are quite common (e.g. López-López et al. 2007, Briggs et al. 2012, Albuquerque et al. 2013), though the success of conservation programmes has also been demonstrated (Devictor et al. 2007, Thomas et al. 2012, Hiley et al. 2013, Smart et al. 2014) and this issue urgently calls for further scientific research. The population dynamics of waterbird species recently driven by environmental and climate changes create new challenges for effective conservation policies and decision making, and must be necessarily based on regular species monitoring. In line with this work, regularly-organised, volunteer-based monitoring should serve as an essential tool in answering our questions about the efficacy of conservation policy.

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Table 1. The list of 28 investigated species in our study

Common Name	Protected	Flyway pop. size	Flyway trend	Index	Latitude	Hunted	Group	Czech estimate	% SPAs
Mute Swan	no	250 000	1	1.74	50.5	no	others	3250	10.4±0.9
Bean Goose	no	592 500	0	32.81	67.3	yes	geese	7500	64.7±6.9
Greater White-fronted Goose	no	1 310 000	1	26.91	70.5	yes	geese	31,550	94.3±1.9
Greylag Goose	no	666 000	1	4.40	54.0	yes	geese	4700	74.1±3.9
Eurasian Wigeon	no	1 800 000	0	5.26	61.5	no	dabbl. duck	220	16.6±2.7
Gadwall	yes	172 500	1	5.28	49.5	no	dabbl. duck	120	37.0±5.4
Common Teal	yes	1 565 000	1	1.34	54.5	no	dabbl. duck	600	29.1±4.6
Mallard	no	4 500 000	0	0.90	53.5	yes	dabbl. duck	179,500	20.6±1.4
Common Pochard	no	1 100 000	−1	6.37	49.0	yes	diving duck	1450	10.4±2.7
Tufted Duck	no	1 800 000	0	2.34	58.0	yes	diving duck	4950	8.2±2.1
Greater Scaup	no	310 000	−1	33.48	65.0	no	diving duck	88	9.5±3.1
Velvet Scoter	no	450 000	−1	27.43	63.5	no	diving duck	66	43.1±9.2
Common Goldeneye	yes	1 350 000	0	2.66	58.5	no	diving duck	1075	14.6±3.2
Smew	yes	75 000	0	16.74	63.0	no	fish-eat	90	46.6±7.1
Goosander	yes	266 000	−1	1.79	60.0	no	fish-eat	3500	20.0±1.8
Little Grebe	yes	405 000	1	4.42	43.3	no	others	650	10.8±0.8
Great Crested Grebe	yes	1,080 000	−1	24.98	48.0	no	fish-eat	280	13.7±4.4
Great Cormorant	yes	392 500	1	1.73	50.0	no	fish-eat	12,050	16.1±1.5
Great White Egret	yes	46 550	1	1.50	45.0	no	fish-eat	780	47.7±2.4
Grey Heron	no	497 000	1	1.79	54.0	no	fish-eat	2600	23.1±0.8
Eurasian Moorhen	no	3 900 000	0	3.34	44.5	no	others	575	48.0±3.8
Eurasian Coot	no	4 250 000	0	1.40	50.5	yes	others	10,750	3.1±0.5
Black-headed Gull	no	5 535 000	0	2.21	54.0	no	gulls	10,100	35.3±7.4
Mew Gull	no	1 850 000	−1	1.82	60.5	no	gulls	2600	13.7±3.3
large gulls	no	2 804 250	1	8.00	53.5	no	gulls	3950	11.7±4.4
White-tailed Sea-eagle	yes	18 000	1	3.48	56.5	no	fish-eat	150	51.6±3.0
Common Kingfisher	yes	239 000	0	2.90	48.5	no	fish-eat	210	14.7±0.5
White-throated Dipper	no	500 000	0	4.48	53.5	no	others	585	9.9±0.4

Notes: Protected – the protection status, Flyway pop. size/trend – Flyway population size/ Flyway population trend (Wetlands International 2015), Index – Water-type specialisation index, Latitude – Latitudinal midpoint (Snow and Perrins 1998, Lemoine et al. 2007), Hunted – hunted species in the Czech Republic, Group – eco-taxonomic group (Snow and Perrins 1998), % SPAs – mean proportions of numbers in SPAs ± SE.

Table 2. The effects of independent variables on proportions of numbers in SPAs (average partial effects, based on a generalized linear model with species-specific random effects).

	(1)	(2)	(3)
Year	0.200***	–0.155***	–0.151***
	(0.000)	(0.000)	(0.000)
Hunting/protection			
– Hunted species	ref.	ref.	ref.
– Non-hunted species	–0.49	–0.84	12.58
	(0.101)	(0.600)	(0.707)
– Protected non-hunted species	–0.49	26.40	44.10
	(0.608)	(0.503)	(0.211)
Log of flyway population size		–0.224	–0.391***
		(0.051)	(0.000)
Flyway population trend		36.20*	–0.261
		(0.039)	(0.932)
Log of Czech population estimate		0.219***	0.215***
		(0.000)	(0.000)
Latitudinal midpoint		6.807**	2.234
		(0.002)	(0.230)
Water-type specialization index		22.60	8.771
		(0.118)	(0.414)
Eco-taxonomic group			
– Dabbling ducks			ref.
– Diving ducks			–0.45*
			(0.040)
– Fish-eaters			–0.67*
			(0.047)
– Geese			77.89*
			(0.013)
– Gulls			39.33
			(0.211)
– Others			–0.40**
			(0.003)
P(group)			0.000
Observations	336	336	336

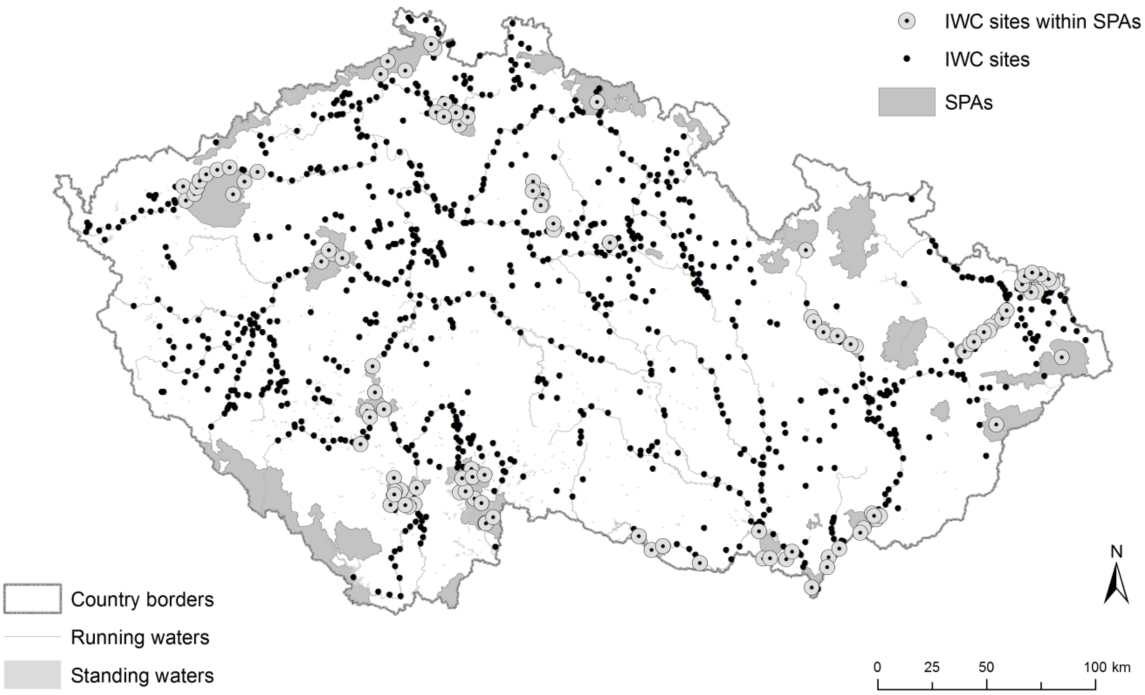
Notes: (i) P-values in parentheses (ii) $^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$.

Table 3. Changes in numbers of 28 investigated species inside and outside SPA network (the overall area is also included)

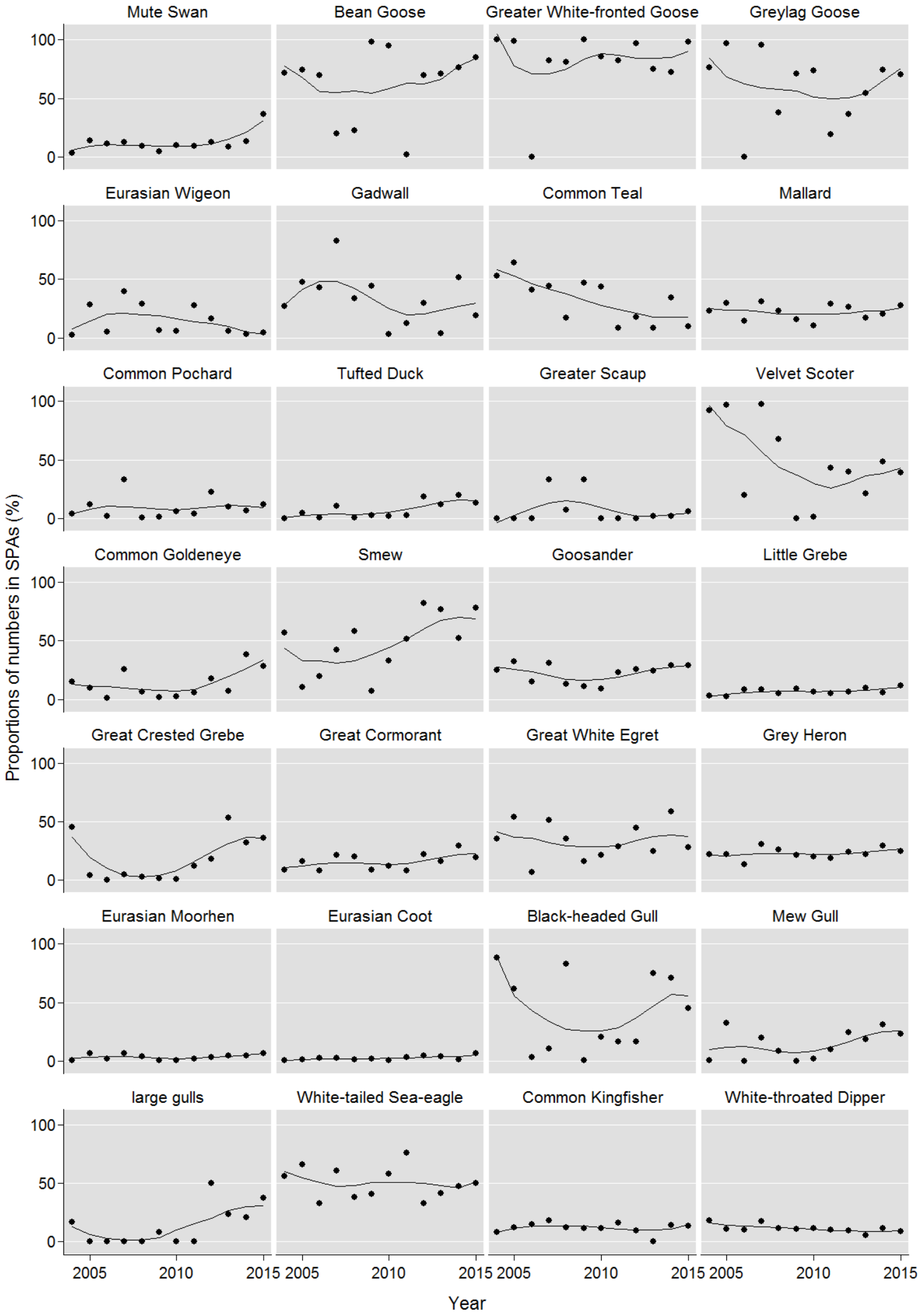
Common name	Inside SPA (overall slope±SE)	trend	Outside SPA (overall slope±SE)	trend	All wetlands (overall slope±SE)	trend	Difference in trends: inside − outside (Wald test)
Mute Swan	-0.023±0.020	U	-0.061±0.001	MD**	-0.056±0.008	MD**	1.53
Bean Goose	0.174±0.188	U	0.061±0.067	U	0.129±0.069	U	0.09
Greater White-fronted Goose	0.262±0.065	SI**	0.477±0.061	U	0.276±0.037	SI**	196.12−−−
Greylag Goose	0.076±0.023	MI**	0.181±0.047	SI**	0.101±0.017	SI**	42.77−−−
Eurasian Wigeon	-0.027±0.052	U	0.061±0.015	MI**	0.053±0.014	MI**	5.82−
Gadwall	-0.013±0.081	U	0.152±0.025	SI**	0.084±0.026	MI**	27.35−−−
Common Teal	-0.144±0.034	SD**	0.053±0.015	MI**	-0.018±0.012	S	63.36−−−
Mallard	0.015±0.011	S	0.007±0.004	MI**	0.009±0.004	MI**	1,65
Common Pochard	0.156±0.078	U	0.082±0.012	SI**	0.083±0.011	SI**	1.51
Tufted Duck	0.269±0.092	SI*	-0.003±0.007	S	0.013±0.007	S	58.22+++
Greater Scaup	0.267±0.220	U	0.228±0.056	SI**	0.226±0.048	SI**	#
Velvet Scoter	0.299±0.168	U	0.147±0.056	MI*	0.178±0.050	SI*	#
Common Goldeneye	0.180±0.031	SI**	0.033±0.010	MI**	0.054±0.009	MI**	42.97+++
Smew	0.129±0.047	MI*	-0.086±0.034	MD**	0.019±0.021	U	26.80+++
Goosander	0.067±0.014	MI**	0.028±0.008	MI**	0.037±0.007	MI**	9,68++
Little Grebe	0.039±0.032	U	0.006±0.008	S	0.008±0.008	S	1,28
Great Crested Grebe	0.422±0.604	U	0.115±0.024	SI**	0.171±0.025	SI**	34.40+++
Great Cormorant	0.038±0.018	MI*	-0.026±0.007	MD**	-0.016±0.007	MD**	13.52++
Great White Egret	0.134±0.024	SI**	0.116±0.013	SI**	0.121±0.011	SI**	1.30
Grey Heron	0.002±0.012	S	-0.024±0.005	MD**	-0.018±0.004	MD**	6.95++
Eurasian Moorhen	0.085±0.054	U	-0.006±0.010	S	-0.004±0.009	S	2.98
Eurasian Coot	0.078±0.029	MI**	-0.003±0.007	S	-0.001±0.006	S	5.54+
Common Gull	0.048±0.1058	U	-0.033±0.027	U	-0.003±0.020	S	24.87−−−
Black-headed Gull	0.218±0.218	U	0.027±0.008	MI**	0.044±0.008	MI	2.96
large gulls	-0.011±0.110	U	0.258±0.034	SI**	0.054±0.016	MI**	57.98−−−
White-tailed Sea-eagle	0.014±0.022	U	0.029±0.016	U	0.021±0.012	S	0.37
Common Kingfisher	-0.020±0.023	U	-0.032±0.010	MD**	-0.030±0.009	MD**	0.11
White-throated Dipper	0.023±0.022	U	0.012±0.007	S	0.013±0.007	S	0.47

Notes: (i) * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. (ii) Categories of trends: SI – strong increase, MI – moderate increase, S – stable, MD – moderate decrease, SD – strong decrease, U – uncertain. (iii) # indicates that the EM algorithm in TRIM failed to converge. (iv) Significance of the difference in trends inside and outside SPA is based on a Wald test and is indicated together with the sign of the difference as follows: +/− $P < 0.05$, ++/−− $P < 0.01$, +++/−−− $P < 0.001$.

Figure 1. Location of investigated wetland sites



680 Figure 2. Trends in proportions of numbers in SPAs in individual species (2004–2015). Trend
681 curves are estimated by LOWESS (locally weighted scatterplot smoothing) with a bandwidth
682 of 0.8.



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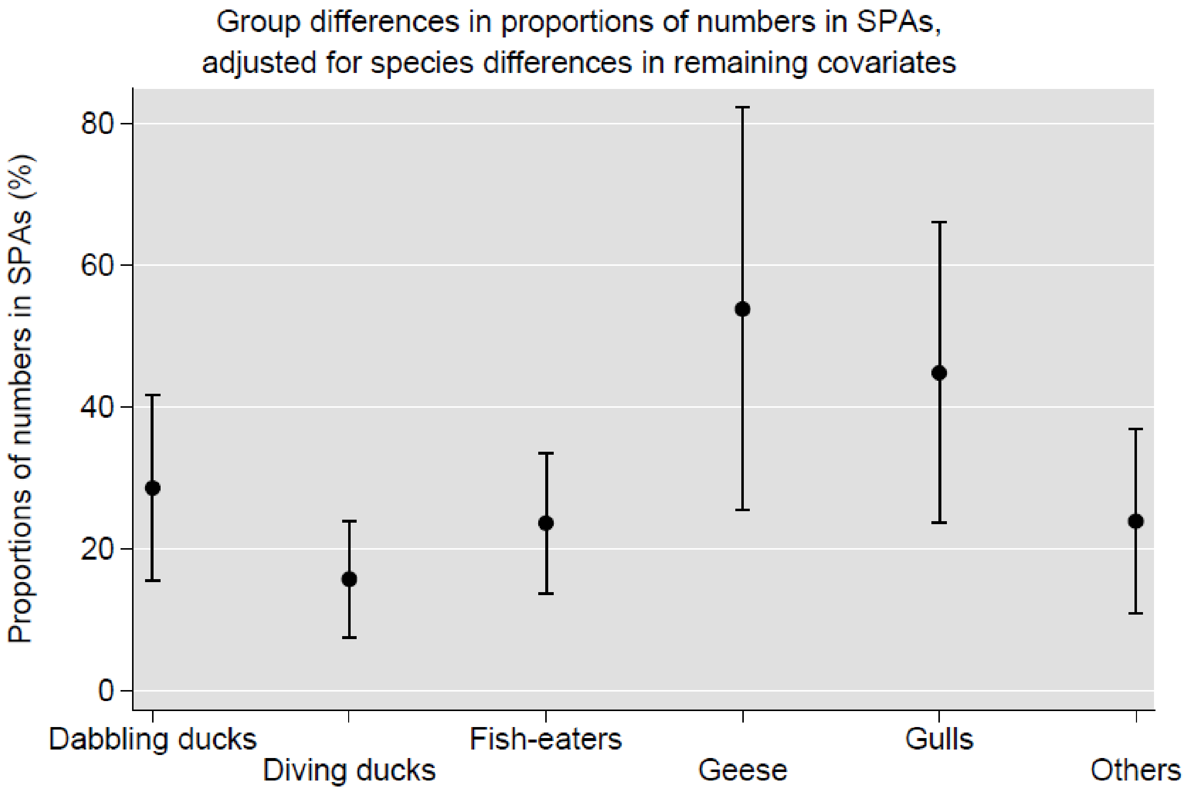
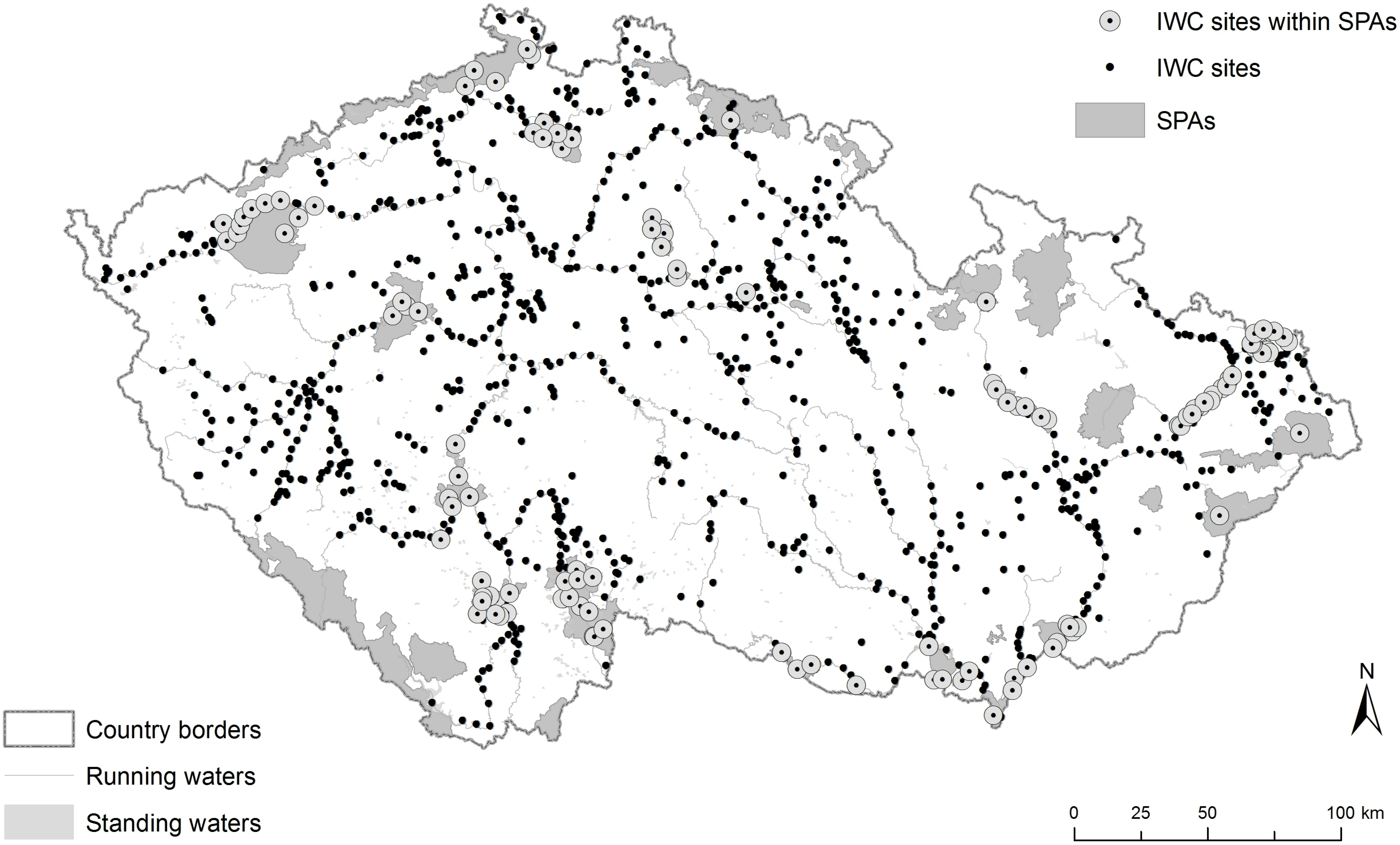
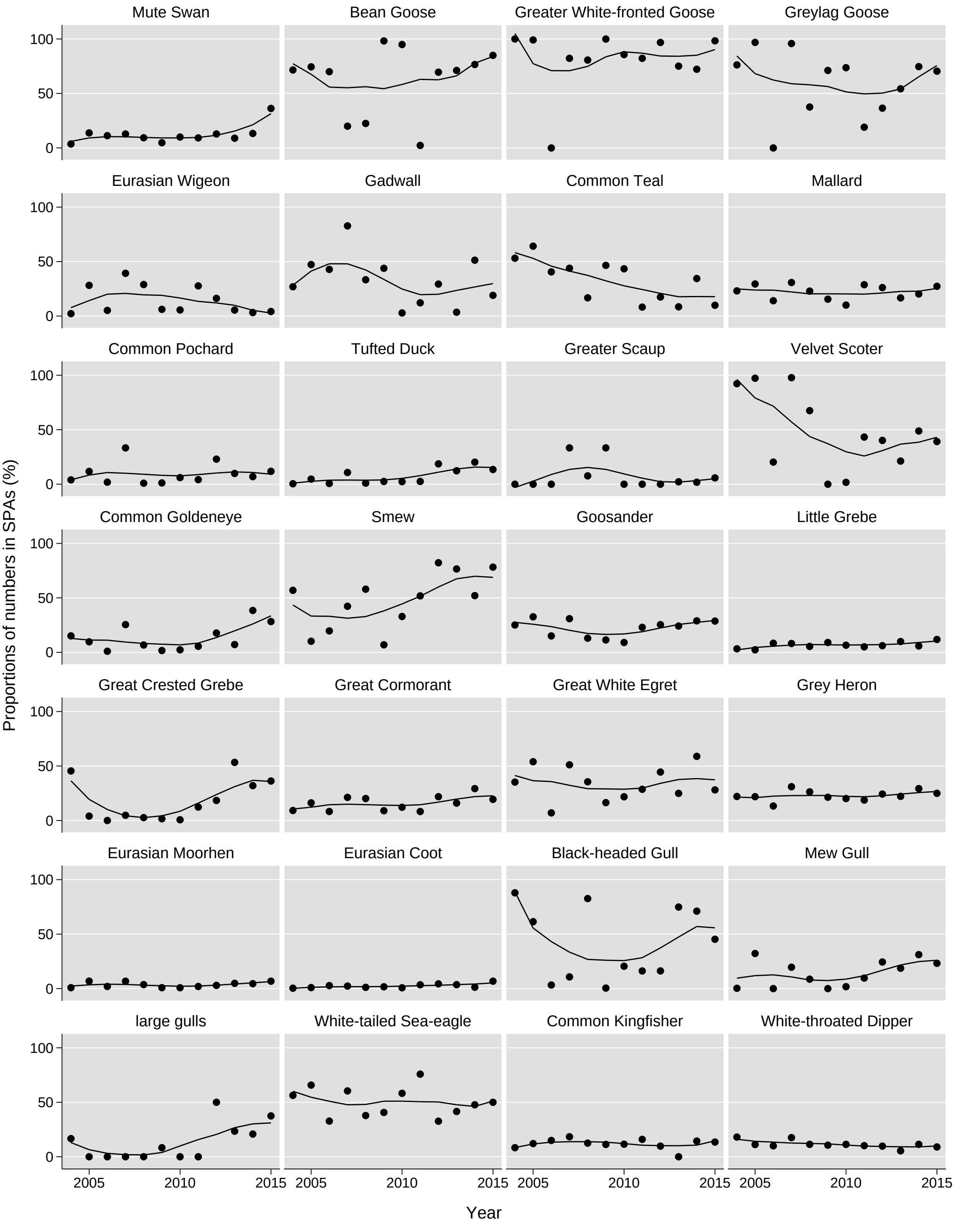
685 Figure 3. Proportions of numbers in SPAs in six eco-taxonomic groups.



686


IWC sites within SPAs
IWC sites
SPAs
Country borders
Running waters
Standing waters
N
0
25
50
100 km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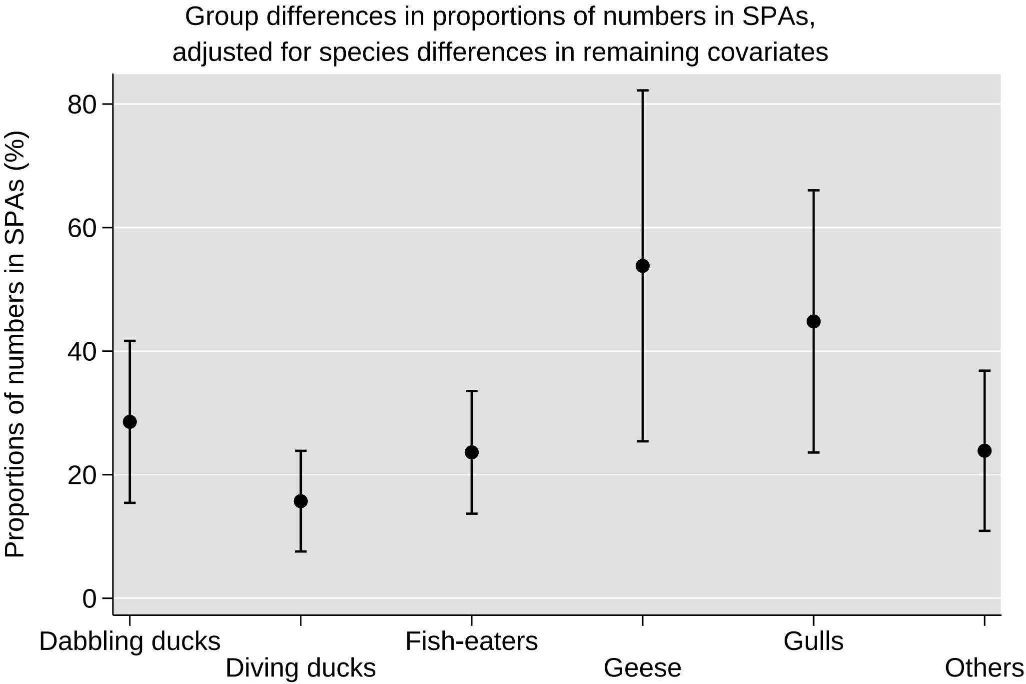
Greater Scaup
Velvet Scoter
Common Goldeneye
Smew
Goosander
Little Grebe
Great Crested Grebe
Great Cormorant
Great White Egret
Grey Heron
Eurasian Moorhen
Eurasian Coot
Black-headed Gull
Mew Gull
large gulls
White-tailed Sea-eagle
Common Kingfisher
White-throated Dipper
100
50
0
2005
2010
2015
Proportions of numbers in SPAs (%)
Year



Group differences in proportions of numbers in SPAs, adjusted for species differences in remaining covariates