Bird Census News

Journal of the European Bird Census Council
www.ebcc.info

2010
Volume 23 n°1-2: Proceedings Bird Numbers 2010 (partim)

Bird Census News
2010, volume 23 n°1-2
ISSN 1381-5261
Download pdf from www.ebcc.info

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Bird Census News reports developments in census and atlas work in Europe, from the local to the continental scale, and provides a forum for discussion on methodological issues.

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SUBSCRIPTION: 2 issues/year: May-June and November-December
Standard rate: 1 year- 2 issues: + individuals: 10 Euro + organisations: 15 Euro
Special offer: 3 years- 6 issues: + individuals: 25 Euro + organisations: 40 Euro

BANK TRANSFER into **IBAN n° NL14 PSTB 0004 2356 70** Postbank Leeuwarden, The Netherlands, **BIC code PSTBNL21** of **EBCC Treasurer** for 'Bird Census News'. Please indicate for which volume (s) you contribute.

Bird Census News is financially supported by the:
Research Institute for Nature and Forest, INBO
Kliniekstraat 25, B-1070 Brussels, Belgium.
The INBO is a scientific institution of the Flemish Community

Bird Numbers 2010. Monitoring, indicators and targets. Proceedings of the 18th Conference of the European Bird Census Council, Cáceres, Spain (partim)

Foreword

This volume of Bird Census News contains 16 additional papers to the Proceedings of the 18th EBCC Conference, *Bird Numbers 2010, monitoring, indicators and targets*, held in March 2010 in Cáceres, Spain, edited by Ana Bermejo of SEO/BirdLife and her editorial team and published as a special issue of Ardeola (volume 57) of December 2010.

The papers in BCN are short notes (only a few pages) as well as longer contributions. I wish to thank all the EBCC Board members and observers for their valuable assistance during the editorial process and/or as a referee of one or more articles: Ruud Foppen, David Noble, Hans-Günther Bauer, Åke Lindström, Henning Heldbjerg, Verena Keller, Lluís Brotons, Oskars Keišs, Mikhail Kalyakin, Petr Voříšek, Mark Eaton, Ian Burfield, Francesc Sardà Palomera and Jana Škorpilova. The EBCC Board thanks Ana Bermejo and the SEO/BirdLife team for their preliminary selection work.

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Please refer to the volume and the article in the following way:
For the volume: Anselin, A. (ed) (2010). Bird Numbers 2010. Monitoring, indicators and targets. Proceedings of the 18th Conference of the European Bird Census Council, Cáceres, Spain (partim). *Bird Census News* 23/1-2.
For an article: Author (2010). Title article. In: Anselin, A. (ed) (2010). Bird Numbers 2010. Monitoring, indicators and targets. Proceedings of the 18th Conference of the European Bird Census Council, Cáceres, Spain (partim). *Bird Census News* 23/1-2: xx-xx

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Towards a new generation of breeding bird Atlases: annual Atlases based on site-occupancy models

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Abstract. Site-occupancy models which take into account detection probability of species have promising applications for Atlas work. These models enable to estimate the 'true' state of occupancy in sites even if sites differ in observation effort. Data collected with different field methods and opportunistic data collected without standardized field methods may be used together to derive reliable inferences on species distribution, colonisation and extinction. This may lead to a new generation of atlases with annual distribution maps on the internet. We emphasize the remarkable perspectives of these models and discuss some problems that need to be solved.

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Introduction

There is a growing need for up-to-date data on the distribution of breeding birds. Such data help in setting priorities in site and species conservation and in measuring the efficacy of management actions. These data may also reveal the impact of large-scale environmental changes on the distribution of breeding birds, such as climate change and land use. Bird atlases with distribution maps per species are therefore considered as important tools in species conservation (Gibbons *et al*. 2007).

Unfortunately, compiling a bird atlas and keeping it up-to-date is far from easy. First, it requires considerable time and effort to get the "complete" picture of the distribution of species in a particular region or country. It usually takes years to complete the field work and to publish the atlas. Repeating the work may be even more challenging as observers might be less inclined to do field work for a next atlas round shortly after an earlier one. Generally, intervals between the publications of successive atlases are growing, and they are becoming increasingly outdated (Gibbons *et al*. 2007). Meanwhile, new data become available but remain unused. As an example, the EBCC Atlas of European breeding birds is already 12 years old, and part of the underlying field work is even much older (Hagemeijer & Blair 1998). Although each year new distributional data become available through the monitoring schemes in many European countries, these data do not result in updated European maps; neither do data from recent national atlases, such as the current atlas for breeding birds in Poland (Sikora *et al*. 2007).

Secondly, many atlases are partly or completely based on opportunistic observations of bird species rather than on observations collected using standardized observation efforts. This means that some grid cells are investigated more often and more thoroughly than others, leading to varying probabilities to detect the presence of a species. This may lead to flawed distribution maps, which reflect observers' preferences and efforts. If e.g. observers prefer sites close to their home or prefer sites in more attractive areas, the presence of species is underestimated in sparsely-populated and less attractive areas. Systematic sampling with standardized observation efforts, such as done for the breeding bird Atlas in the Netherlands (SOVON 2002) reduces this bias, but requires much larger efforts, which may not be feasible in countries with few observers.

Therefore, the dilemma in composing a breeding bird atlas is either to invest considerable amounts of time, effort and money to make a reliable up-to-date atlas or to be content with a less timely and less reliable product. The recently developed site-occupancy models (MacKenzie *et al*. 2006; Royle & Dorazio 2008) may offer a way out. These models enable to create a distribution map even if not all constituent grid cells of the map have been surveyed. Moreover, they can adjust for differences in detection probability, allowing opportunistic data to be included without causing too many difficulties. For these reasons, we believe that future distribution maps may be compiled on an annual basis. We briefly describe the site-occupancy model and then outline the data and analyses needed for the model. We also speculate about the opportunities to compose an annual European atlas.

Site-occupancy model

Site-occupancy models are models to analyse presence-absence data taking into account imperfect detection of species (MacKenzie *et al*. 2006; Royle & Dorazio 2008). These models require data of temporal and spatial replicates, i.e., for at least a number of sites repeated surveys are needed within the breeding season (technically speaking: within a species-specific closure period, which means that a site must stay either occupied or not by the species but must not become permanently abandoned or colonised during the period of surveys). The data must be arranged in so-called detection histories per site (here, grid cell) during a single season. An example is "0-1-0" for a species detected during the second visit, but not during the first and third visit to a site in the breeding season.

These models are useful to analyse presence-absence data collected with a standardized field protocol (Royle & Dorazio 2008), but they can also be applied to opportunistic data, preferably (see below) data in the form of daily species lists (Kéry *et al*. 2010; Van Strien *et al.* 2010). The basic idea behind their use for opportunistic data is that variation in observation effort over the years is directly translated into variation in species detectability. Replicated visits to a site are required to allow estimating the probability of detection separately from the probability of occurrence (Kéry *et al*. 2010). Taking into account detection probability in the analysis thus relaxes the need to standardize field method and effort. The model estimates the occupancy for all sites together (i.e., the proportion of occupied sites in the statistical population), but also the probability of occupancy ("true" state) for each individual site. These estimates of occurrence (true presence and true absence) per site enable to create a distribution map.

Furthermore, data of multiple years can be analysed by dynamic site-occupancy model such as described by Royle & Dorazio (2008). These models describe annual occupancy in each site in year t as a function of occupancy in year t-1 and year-specific colonisation and extinction (= 1-survival) rates. In other words, the dynamic model is an extended metapopulation model to estimate probabilities of occupancy, colonisation and extinction, corrected for imperfect detection. Both occupancy/colonisation /extinction and detection probability may be formulated as a function of covariates. E.g. if one expects that detection probability differs between monitoring data and opportunistic data, the data source may be included as covariate (Van Strien *et al.* 2010).

Data sources

A species distribution map should provide information about the presence or absence in all constituent grid cells of the map. But reliable recording a species' distribution does not necessarily imply the need to survey each grid cell or to survey the same grid cells every year. The more grid cells are being surveyed, the more precise the predictions are of course, but a representative sample may be sufficient. Presence or absence in non-surveyed grid cells may then be derived from the information in the surveyed sites. Existing spatial analysis modelling methods do this too and estimate the probability of occupancy in non-surveyed grid cells based on regression analysis, using covariates such as habitat and region, or based on spatial autocorrelation, or on a combination of both approaches (see e.g. Elith *et al*. 2006). But the dynamic site-occupancy model also takes into account detection probability as well as temporal autocorrelation in occupancy (Royle & Dorazio 2008). This makes it possible to achieve annual maps even if many sites were not surveyed in each year.

Because the maps may in principle be composed from sampling data only, we consider the data from national bird monitoring schemes as important sources of data for the models, for the following reasons. The monitoring data are typically annual observations collected in many sites; so many spatial replicates are available. Monitoring sites are usually widely distributed and often (stratified) randomly selected across the country, ensuring an appropriate coverage of the country. This is frequently not the case for opportunistic data, where data for some regions may be entirely lacking. Moreover, many monitoring protocols include repeated visits which are a prerequisite of the site-occupancy models. Being count data of birds, the data need to be converted into presence-absence data per grid cell.

Monitoring data alone however are often too sparse to produce a detailed atlas, so opportunistic data are essential as a second source of data for the models. These data may be single species records on a particular date and site, without information about any other species. Such data are usually coincidental observations and are predominant in museum collection data. Opportunistic data may also be records of several species made by one observer on one particular date and site, but the most useful data are daily species lists with records of all species seen or heard, because these are also informative about species not detected (see below). Opportunistic data are easier to collect than standardized monitoring data and currently an increasing amount of data is being collected in the framework of citizen science projects with easy data entry facilities on the internet (e.g. telmee.nl, waarneming.nl, ebird.org, worldbirds.org).

A site-occupancy model assesses the annual "true" state for each individual grid cell (e.g. 5 x 5 km square) from monitoring as well as opportunistic data (Van Strien *et al*. 2010). Site-occupancy models however cannot work with presence-only data, so non-detection records need to be deduced from the data prior to the analysis. This is straightforward in the case of monitoring data and complete daily species lists, because any species not recorded may be considered as not detected at that date and site. Non-detections may also be deduced for single species records and incomplete daily lists. But non-detections then also include the non-reporting of detected species, which lowers the usefulness of the latter two data types. Van Strien *et al*. (2010) found that single species records and incomplete daily lists of Dutch dragonflies produced much less precise occupancy estimates as compared to comprehensive species lists.

A third source of data is formed by atlas studies in a particular region or country. These studies may either cover all species or may be dedicated to one or a few species. If repeated visits were collected in at least a subset of sites, in principle these data can also be used for site-occupancy models.

Running a site-occupancy model

Site-occupancy models may be fitted using the programs PRESENCE or MARK (see MacKenzie *et al*. 2006). The implementation of the models in these programs is based on maximum-likelihood estimation and computations do not require much time. Regrettably, so far these programs do not allow large datasets and elaborated models. Also both programs are designed for running one model at a time and not for analysing a large number of models for many separate species in batch mode.

Alternatively, models may be fitted in a Bayesian mode of inference using the WinBUGS software (Spiegelhalter *et al*. 2003). WinBUGS can handle much larger datasets and is more flexible in model choice. Models may be fitted for a number of models in batch-mode from R, using the library R2WinBUGS. But the time needed to fit an elaborate model with many sites easily grows to hours before model convergence is achieved. This is not practical if many species and many different models need to be processed on a routine basis. We expect however that processing time can be reduced in the future, among others by the use of informative priors derived from earlier runs which may speed up model convergence.

Opportunities for an annual European Atlas

Apart from computational problems, there are some theoretical problems that deserve attention in the future, such as the exact procedure to create zero values from opportunistic data (Kéry *et al*. 2010; Van Strien *et al.* 2010) and the heterogeneity of detection probability across sites (see MacKenzie *et al.* 2006; Royle & Dorazio 2008). Despite such problems, Van Strien *et al.* (in press) achieved credible annual distribution maps for the Netherlands for a butterfly species based on opportunistic data only. Therefore, we are rather optimistic about the opportunities to compose annual distribution atlases at a national scale in the near future.

An annual atlas at European scale is more challenging. Such an atlas could be built upon the many national monitoring schemes in Europe, as well as on the growing amount of opportunistic data. In some countries opportunistic data might well be the only information on species' distribution one can collect in practice, because observers are few. Even then, the amount of opportunistic data in these countries might remain too poor to achieve sufficient coverage of the country. An option is to collect replicated daily species lists in a limited number of randomly selected grid cells in countries where standard monitoring is not possible and opportunistic data are scarce.

Field methods used in large-scale monitoring schemes differ between countries (see www.ebcc.info) and thereby the probability to detect species in a site if present. But site-occupancy models can handle such differences by their ability to adjust for differences in detection probability. When using site-occupancy models it is therefore not needed to harmonise field methods between countries, to attune observation efforts between countries or to calibrate data between countries, in order to derive proper inferences on supranational distribution of species and changes therein.

It is impractical to include all data of a species from all countries in one site-occupancy model because the number of grid cells would be too large to be processed in a reasonably amount of time. It is better to run separate models for each country, or parts of countries, and then to collate the results (Figure 1). Supranational collation is straightforward because when for each grid cell the 'true' occupancy is estimated, country maps can simply be "glued" together to create a European map (although estimating a supranational trend and its confidence intervals from collated information is not straightforward). When models are run for individual countries, grid cells sizes do not necessarily need to be equal between countries. If more fine-scaled data are available, a country might want to use a smaller grid size than elsewhere. But for some applications, such as assessing changes in the European range of species it is required to standardize grid cell size and to use, say a 25x25 km^2 or perhaps even a 10x10 km^2 resolution. The latter corresponds to the resolution required for reporting the range of species under the Habitat Directive and which perhaps becomes an obligation under the Birds Directive too.

One way to run an annual European atlas is to store all data from all countries in one database, after which a central coordination unit runs the model per country and then collates the country results to produce the supranational species maps. This would require proper arrangements concerning data ownership and use. Another way would be similar to the procedure adopted in the Pan-European common bird monitoring project (PECBMS), where national organisations deliver their national indices produced by the TRIM program instead of their raw data (Gregory *et al*. 2005). This would mean that national organisations run the site-occupancy model themselves for their own data and deliver occupancy estimates to a central coordination unit. But site-occupancy models are less easy to run than TRIM. Apart from the need to develop user-friendly programs it then requires considerable efforts to train the participants.

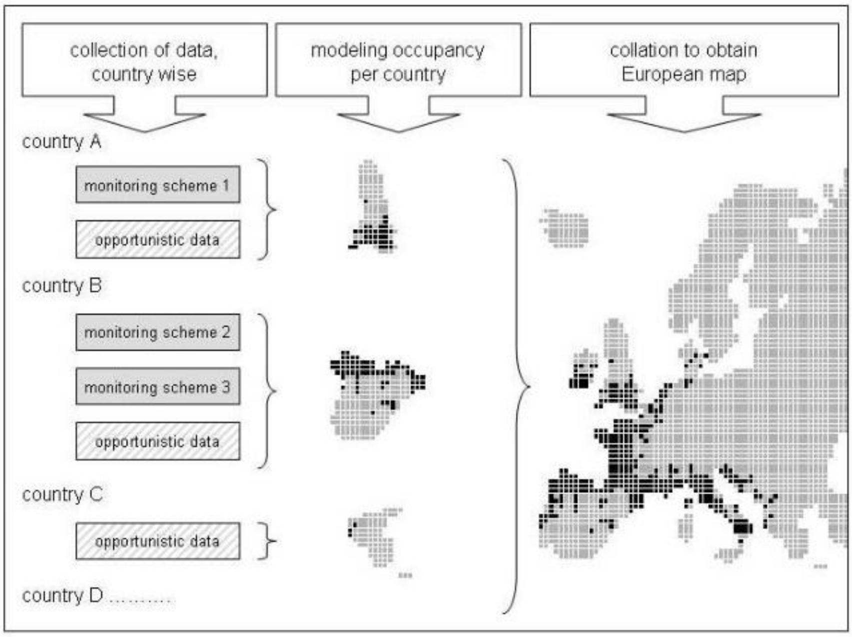

Figure. 1. Composing an annual European Atlas using site-occupancy models per country.

Perspectives

Site-occupancy models may lead to a new generation of atlases: annual distribution maps on the internet, with colonisation and extinction indications per grid cell. The site-occupancy models are expected to produce reliable inferences on distribution (presence/absence) and trends in distribution. More specifically they may produce: (1) the actual number of occupied grid cells in Europe (species range), and trends therein,(2) colonisation and extinction rates of species and (3) the changes in occupancy per species per grid cell (trend maps). It is also possible to combine the maps of individual species to produce (4) the total number of all breeding species, or subsets of species groups according to habitat or trait per grid cell and trends therein. Importantly, for all parameters confidence intervals of the estimates are available.

The expected results from a European atlas open new perspectives for conservation and research. They allow estimating the actual European range of breeding birds, which is an important parameter in evaluating the EU's Birds Directive. Furthermore, the atlas facilitates studies on the impact of climate change on range shifts (Lepetz *et al*. 2009) and make it possible to track if the actual changes match those predicted by the climatic Atlas (Huntley *et al*. 2007). They can be of help to make the European policy more climate change-proof, e.g. by facilitating to examine if Natura-2000 sites are useful or can be made useful to support the colonisation of species that need to alter their distribution to keep track with suitable climate conditions. They can also be used to identify the best locations for future Natura-2000 sites or to track changes in High Nature Value farmland areas. The annual results, together with population monitoring data, also allow a frequent and statistically sound update of the red list status of many bird species which is useful for a Red list indicator of European birds. And so on and so further. More generally, site-occupancy models using monitoring and opportunistic data can be an important ingredient of a systematic program to determine the effectiveness of the Birds Directive and other European Union's policy instruments. But perhaps above all, annual distribution maps on the internet, with colonisation and extinction indications per grid cell are a great communication tool because it is very appealing for the general public to see almost real-time changes in bird population happening in relation to climate change, land use and conservation measures.

Acknowledgements

Marc Kéry has inspired us to explore the possibilities of site-occupancy models. Willy van Strien gave helpful comments on an earlier version of the manuscript.

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Received: 11.05.2010
Accepted: 28.01.2011

Bird Census News 2010, 23/1-2: 9-20

Checklist programs as a source of data for bird monitoring: designing analyses and model validations to account for unequal spatial and temporal sampling effort

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Abstract. Data on bird distribution and abundance are being collected in an ever-increasing number of countries using internet-based data collection schemes. We work with data from one well-established checklist schemes in North America, called eBird. Our experience with constructing species distribution models from these data has shown that an important challenge in analysis is dealing with data that are not evenly distributed across the continent or among times of year. Here we summarize recent work in developing a novel method for creating and validating species distribution models from bird checklist data, producing predictions of distributions throughout the year.

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Introduction

Large amounts of data are needed in order to describing the distributions of birds and identifying habitat associations, as well as assess status and trend of bird species across their ranges. Given the challenges of collecting the needed data, there is growing interest in using the records of bird watchers — bird watchers' checklists — as a source of information. The knowledge that these checklist data contain useful information has spurred the growing number of European regions to start internet-based collection of these data (e.g., U.K. and Ireland: BirdTrack; Switzerland: Swiss Avian Information Service; Italy: ornitho.it; Portugal: PortugalAves). Our experiences in North America are that these observational, checklist data can be collected in very large volumes and at extremely low cost per observation once such checklist schemes are well established (Sullivan *et al*. 2009).

The use of data from checklists is not without challenges, however. These data are not collected using a carefully designed protocol, and thus an array of biases can distort the relationship between observations and actual biology. Existing analysis methods can account for some of these biases (Kéry *et a*l. in press), whereas other sources of bias may require special adaptation of existing methods. Variation in the density of available data (Figure 1) is one source of bias for which explicit correction may be required. Figure 1 shows the locations of all checklists submitted to the eBird checklist scheme in a large part of North America in a single year. The most obvious pattern in this figure is that more data come from areas with larger human populations: the densely populated northern and central Atlantic coastal region of the United States, and other major metropolitan areas such as Chicago, Salt Lake City and San Francisco are clearly visible.

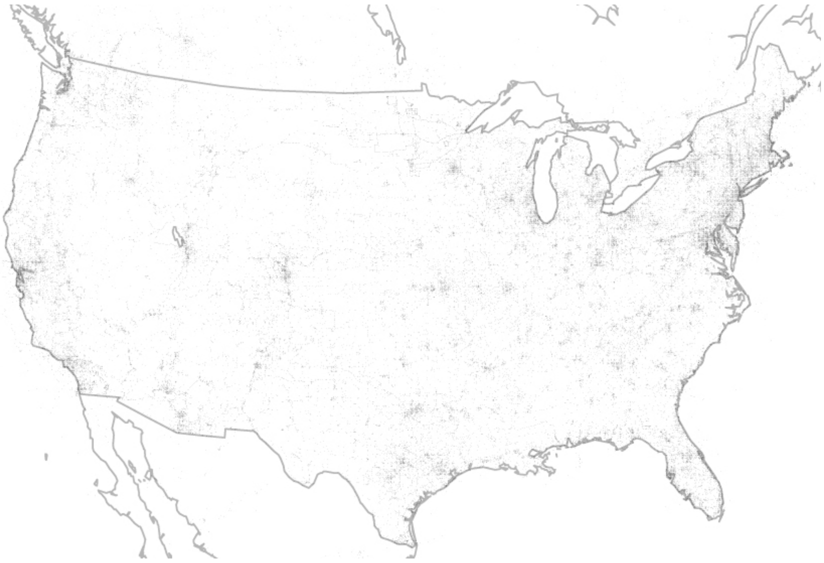

Figure 1. The locations of all checklists submitted to the eBird checklist scheme in 2008. Note that several checklists are often submitted for an individual location, so that this map is not an entirely accurate estimate of the variation in density of data, but only of densities of locations.

Uneven densities of data are a potential problem for two aspects of analyses: creation of realistic models, and validation of these models. The models created during data analysis will be most influenced by regions from which larger amounts of data are available. For example, logistic regression (for creating models of presence or absence of birds) will typically make better predictions for the most common response (present, or absent; Hosmer & Lemeshow 2000); thus for a species whose distribution is mostly in an area of low data density (e.g., grassland-nesting birds in the centre of North America) models of habitat preference might be relatively uninformative because they would be least accurate for regions where the birds were actually present. Similarly, standard assessments of the accuracy of models will also be weighted most towards regions of higher density of data.

The problems caused by uneven densities of data are compounded when attempting to model changes in distribution over time, such as migrations, because data are likely to be unevenly distributed both through space and time. Checklist schemes provide data from throughout the year, unlike most other sources of data on bird distributions. Thus the potential exists to create models of seasonal dynamics from these data, as do the problems associated with data unevenly distributed through both space and time. Unless variation in density of data is explicitly taken into account, we will not obtain the results that we likely want: models that are of as high accuracy as possible across all regions and times of year for which data were analysed.

In this paper, we discuss a novel approach to the analysis and validation of checklist data, with the goal of creating accurate models of species distributions and accurate insights into habitat preferences of bird species. Because our source of data, eBird (Sullivan *et al.* 2009), gathers information throughout the year, we have worked to develop analytical techniques in which the distribution models that are created will describe not just distributions in space but also how these distributions changes through time. Our goal is to create a single model that simultaneously describes differences in distribution through space and time, instead of being forced to create separate distribution models for each arbitrary slice of time. Below, we first discuss our choice of the general family of analysis techniques. Then, separately for the steps of model creation and model validation, we illustrate the problems with using existing methods and suggest solutions. Finally, we discuss the broader relevance of our experiences.

Exploratory analysis methods for species distribution modelling

Model creation

Very early in the process of learning how best to analyse checklist data, we decided that our techniques for analysis would need to be exploratory, as opposed to confirmatory. In other words, we assumed that for most species of birds, we would have insufficient prior knowledge to be able to create a good parametric description of the factors that affect a bird species across its entire range and for all times of the year. Failure to specify an accurate parametric model can lead to extremely erroneous conclusions even for analyses that diagnostic statistics suggest are good (Fitzpatrick *et al*. 2009). This led us to use analytical tools developed in the computer sciences field of machine learning, rather than parametric statistics. Hochachka *et al*. (2007) describe the philosophical and functional differences between these exploratory, “non-parametric” analysis techniques and the parametric statistical techniques with which ornithologists are more generally familiar. The most important difference between parametric statistical techniques and the non-parametric techniques that we are using is that much of the structure of the models — the choice important predictors of distribution, and the forms of relationships between predictors and response (i.e. linear, quadratic, or highly complex curves) — is found as part of the non-parametric analysis instead of having to be specified in advance.

Another early decision that we made was to produce models of distribution and not abundance: we are predicting variation in the probability of presence and not variation in indices of abundance. In analogy to parametric statistical analyses, we are using logistic regressions, and not linear or Poisson regressions. We made this decision because the past experience by our group and others has been that creating presence/absence models is a simpler problem using non-parametric analysis techniques, and we wanted to minimize the complications inherent in our initial exploration of the uses of bird checklist data.

The specific analysis technique that we have used is a form of bagged decision trees. “Bagging” (Bootstrap AGGregatING; (Breiman 1996) is a general technique for removing bias from model estimates. The process involves taking a bootstrapped sample of data from the original data set, and then fitting a model to this bootrapped sample and calculating predictions (such as presence or absence of a species of bird at a set of locations). The process of bootstrapping and calculating predictions is repeated, conventionally several hundred times, and the “bagged predictions” are the arithmetic averages of all predictions for a given location. The predictions from each individual bootstrap sample may be highly biased, because they are the result of a model that fits both the biological signal and the random sampling noise in the data; averaging across the individual models' predictions removes bias because the biases are not consistent from sample to sample. In our work, we created the individual models using decision trees (see (Hochachka *et al*. 2007); however, any base model type (e.g., linear regression, generalized linear mixed models, decision trees) could be used. Our choice of bagged decision trees was based on past experience with the method, which requires little "tuning" of model-creation parameters in order to create accurate models. Bagged decision trees typically just work reasonably well without any special intervention and tuning from the data analyst (Caruana *et al*. 2006). This suited out ultimate goal of automating the process of producing distribution maps for a large number of North American bird species. Several other non-parametric methods exist (see e.g., De'ath 2007, Elith *et al*. 2008) that may be equally or more accurate when the data analyst can take the time to adjust modelling parameters for individual species.

Assessing model fit

Our methods for assessing the validity of species distribution models are also note widely used by ornithologists, who are more familiar with use of p-values, confidence limits or AIC scores to assess the appropriateness of models. Instead, we have used cross-validation. With cross validation, part of the data ("testing data") is selected in advance and not used in the analysis that creates the species distribution model; predicted responses are then calculated for the testing data using the model. A good model is one that can accurately predict observations that are outside of the data used to create the model (i.e. the testing data). Several measures of model fit are available, with the area under the curve (AUC) statistic from receiver operating characteristic curves (Hosmer & Lemeshow 2000) being one measure that is commonly used for models of presence/absence data.

One reason for using cross-validation in order to assess models is inherent in our method of analysis: there is no way of extracting the information needed in order to calculate a p-value or AIC score. Calculation of confidence intervals is possible with bagged decision trees but impractical, because confidence intervals could only be calculated by bootstrapping. Two levels of bootstrapping would thus exist: bootstrapping within the creation of each bagged decision tree, and an outer layer of bootstrapping for the construction of multiple bagged decision trees. Calculations would require extremely large amount of time, measured in days or weeks of processing time for even a single species distribution model.

Cross-validation is not just a poor alternative because other methods of model assessment are not available. Cross-validation also has two useful characteristics that are not shared by p-values, confidence limits or AIC scores. First, cross-validation directly measures a model's ability to extrapolate beyond the data used to create the model, and this sort of extrapolation is typically the intended use of a model. In contrast, p-values and confidence limits are indicators of model fit that are only presumed to indicate how well the patterns in a model will extrapolate. The basis for utility of AIC scores is that they approximate a form of cross validation (Burnham & Anderson 2002), but only asymptotically (i.e. with very large sample sizes). The second useful characteristic of cross-validation is that is provides an objective measure of how far below perfection a model falls. A perfect model would be able to perfectly predict the observed data in the testing data set, and the distance below this perfect state is measured. In contrast p-values and confidence limits will only really tell one whether a model is better than chance alone, but do not provide any clear information about how far from perfection "better than chance" really is. AIC scores provide only a relative and not an absolute measure of the fit of a model, a measure that is relative to the best model in a set. However, even the best model in a set can be an extremely poor model.

Building species distribution models

Standard analysis techniques used with uneven data density

A standard bagged decision tree analysis is not guaranteed to recognize that observations from very different locations and very different time periods might be the result of different habitat associations. An extreme example of the problems that can result is shown in Figure 2. This map shows the distribution of Tree Swallows (*Tachycineta bicolor*) that is predicted by a standard bagged decision tree for January, using data from the entire year in order to build the model. Note in particular the area of bright colour, indicating high predicted prevalence of the swallows, in the very north of the central United States. There are no living Tree Swallows in this area in winter, because the region has average daytime temperatures below freezing and no available food for swallows in January. The data used to create this model do not have any observations of Tree Swallows being present in January. This is clearly an error.

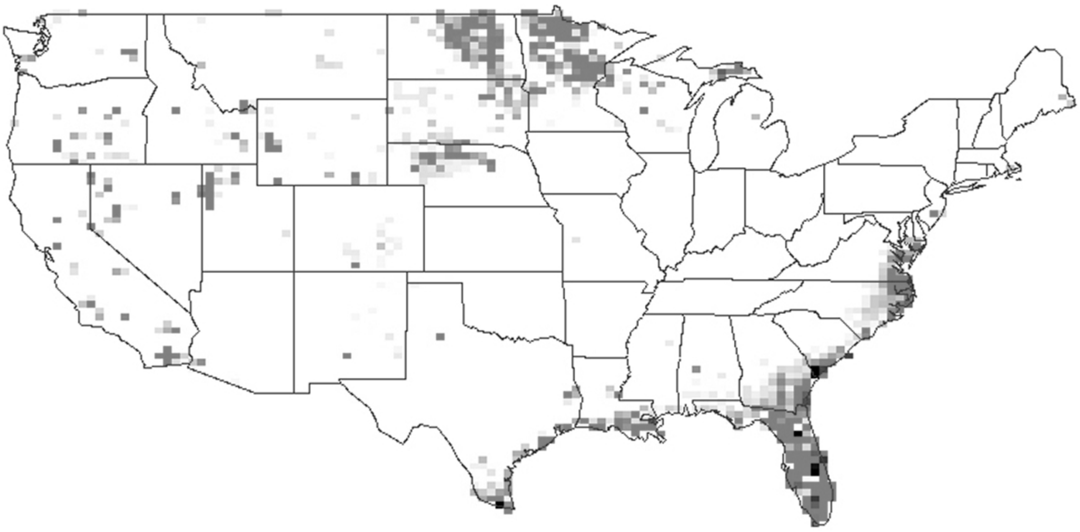

Figure 2: The predicted distribution of Tree Swallows *Tachycineta bicolor* in the United States in January of 2008. These predictions were made by a model of Tree Swallow distribution created using a standard bagged decision tree analysis. Lighter colours indicate higher predicted prevalences of Tree Swallows. For the analysis that created this map, data from throughout the year were were. Date was one of the predictor variables used, so that the analysis should have been able to identify that the northern part of the United States, away from the oceans, does not have Tree Swallow present in winter. However, the model predicted that an area in the north-central United States (in the states of North Dakota and Minnesota) would commonly host Tree Swallows in winter. Although winter data from this region were sparse, there were no data to indicate that Tree Swallows should be present in this region in the middle of winter.

This mistake was the direct result of the uneven distribution of data provided in eBird. We believe that two errors occurred. First, the north-central United States does contain habitat that is highly suitable for Tree Swallows in summer, and more data are available from summer than winter in this region. As a result, we believe that the greater volume of summer information indicating suitable habitat in the region “bled through” the seasons into winter, because there were insufficient data from the winter to counter-balance. Note that this problem occurs with these data, even when the date and monthly mean temperature were predictor variables in the model, so that the analysis should have been able to learn that Tree Swallows cannot live some place at times of year when temperatures are below freezing. The second error is the result of the our habitat predictors being of relatively coarse resolution; the same habitat type that is common in the north central United States is also in southern Florida (the far south-eastern United States), where Tree Swallows are abundant in winter. We also have a higher density of observation in winter from Florida than in the north-central United States. Thus, we believe that the high density of observations in Florida in winter caused the winter habitat association in Florida to erroneously “bleed through” to the north-central United States.

Spatio-temporal Exploratory Models

Clearly, in order to produce accurate species distribution models we needed to find some way to prevent information from being shared between locations that were far apart in time or in distance. Because we wanted accurate distribution models to be produced with little or no special intervention by the data analyst, we did not want to have to create special and arbitrary divides in the data. For example, we did not want to have to create entirely separate models for summer and winter, or models from northern and southern locations where we would have to first divide the data and only then conduct out analyses. Instead, we wanted to create a method that would separate data from distant locations and distant time periods without the analyst having to make species-by-species decisions about the specific separations.

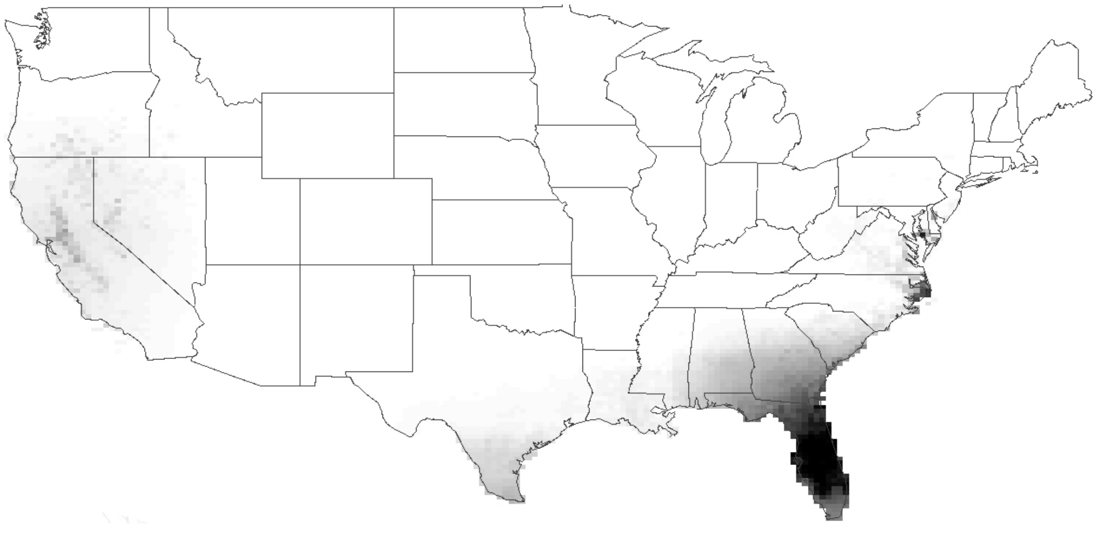

Figure 3: Predicted distributions of Tree Swallows in the United States in mid- to late January of 2008, from an analysis technique that does not allow effects of predictor variables to influence regions too distant in space or time from the locations of individual predictions. Note how this map accurately indicates that no Tree Swallows are found in the north-central United States in the middle winter, in contract to the predictions showing in Figure 2. A Spatio-temporal exploratory model (STEM) was used to produce this map.

The method that we developed to do this, which we call a Spatio-Temporal Exploratory Model (STEM), achieves out goal of ease of use and produces more accurate models of species' distributions (Figure 3). The predicted distribution map in Figure 3 is for the same species and month as in Figure 2, but the STEM analysis accurately shows the absence of Tree Swallows from the north-central United States.

A STEM analysis works by building up the entire model from a large number of sub-models, each built using only data from a relatively small spatial area and relatively short time period. These sub-models overlap, and every data point is used in a large number of sub-models that cover different time periods and spatial regions (Figure 4). For any specific location and date, the predicted probability of a species being present is the average of the predictions from each individual sub-model that was build using data from that location and date. The result is a smooth blending and combining of the sub-models, while at the same time it is impossible for information from summer to influence predictions in winter or information from Florida to influence predictions in North Dakota. Details of the method are described in Fink et al. (in press).

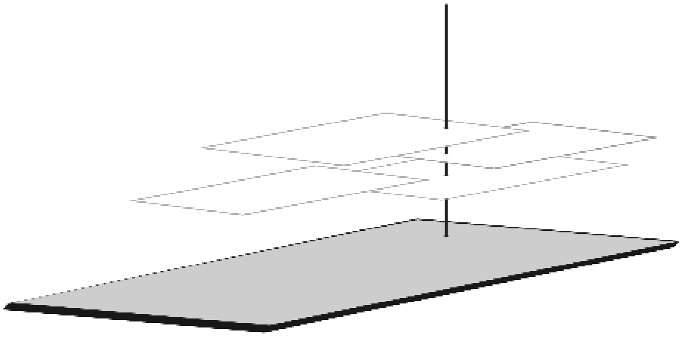

Figure 4: Simplified schematic representation of averaging across multiple sub-models in a spatio-temporal exploratory (STEM) analysis. If the region over which you want to predict a species' distribution is represented by the grey surface at the bottom, a series of partially overlapping sub-regions (white regions above the surface) are chosen and within each sub-region a separate species distribution sub-model is created. The final prediction of a bird's presence at any location is calculated as the average of all of the predictions of sub-models that would contain this location (here represented by the 3 sub-regions through which the vertical line passes). In reality, each location will be contained within a larger number of sub-regions. Further, this schematic represents a single period of time; the true process is repeated for a large series of partially overlapping time intervals.

Validating species distribution models

Conventional cross-validation techniques (selection of testing data)

The choice of data to withhold from analysis, the testing data, can dramatically affect the assessment of model fit in cross validation. Figure 5 shows how dramatically different conclusions can be reached depending on the choice of testing data. One conventional way to select the testing data is to randomly select the lines of data in the data file to withhold. This likely will not be appropriate when analysing bird checklist data, because multiple observations are often made at many locations. If data from some locations are found in both the data used to build the model and the testing data, the testing data are not a completely independent set of data, and cross-validation will likely over-estimate the predictive power of a model. In order to avoid this problem, one can randomly select locations from which all data go into the testing set. While this second approach will avoid the most direct problems of non-independence, the locations used for testing data will still likely be close to and intermixed with the data used to construct the model. While technically speaking, these nearby observations may be statistically independent (Schooley 2006), they are still likely from very similar environments.

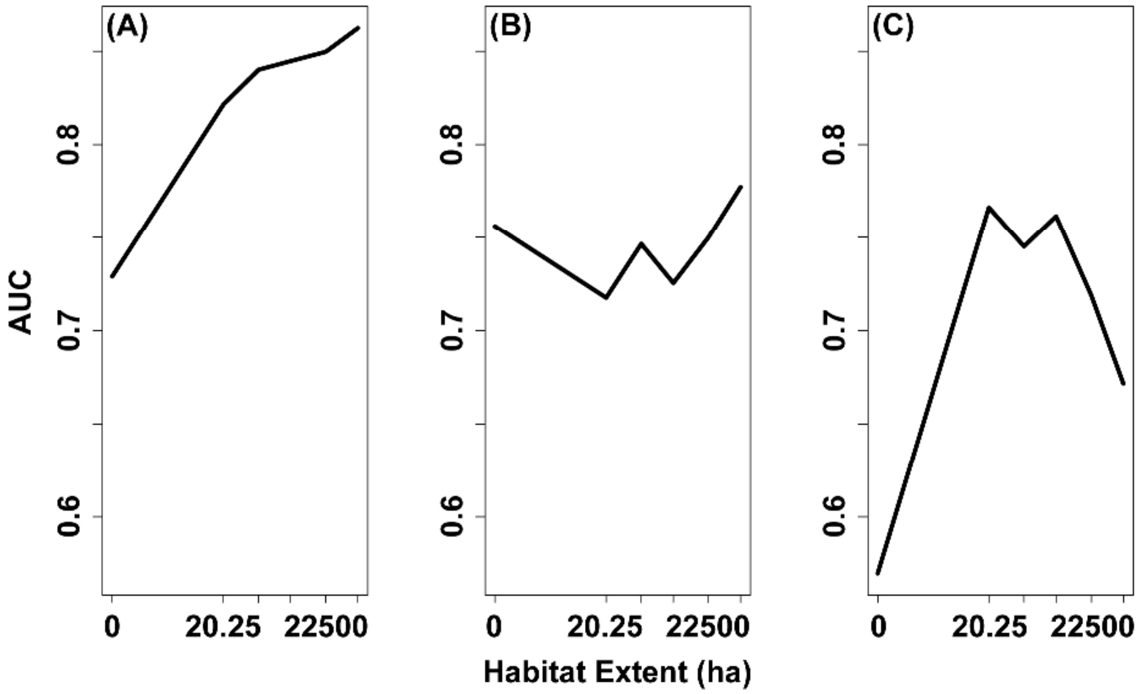

Figure 5: The choice of hold-out (testing) data for cross validation can affect conclusions. Each panel in this figure shows the results of cross-validation for 6 models. Each of the models predicts habitat distribution of one bird species, the Red-winged Blackbird *Agelaius phoeniceus*, based on a different set of potential predictor variables: descriptions of habitat in different areas around the locations of observations (moving from smallest to largest area as you look from left to right). Results from each panel came from the same set of data. The AUC statistic is used to assess model fit; larger AUC values indicate better fit. Lines are used to connect the AUC values for the six models in order to help visualize trends in model fit across the spatial extents of the habitat data. The objective of the analyses was to determine the most appropriate scale at which to describe habitat when predicting distribution. In (A) the testing data were randomly selected from all available data. In (B) a random set of locations was chosen as the testing data; there were multiple observations at each location. In (C) all data from entire sub-regions were withheld for testing purposes. Qualitatively different conclusions would be reached about which model was best, depending on the choice of testing data.

If the goal of analysis is to create a model that extrapolates into areas without data, then the testing data set should be selected so that the performance of longer-distance extrapolation is assessed. Thus, a third approach is sometimes used (e.g., Brotons *et al*. 2004), in which a landscape is gridded into a checker board (Figure 6).

Figure 6: Creating a checker board grid to separate the data used for model construction from testing data will result in model cross validations with spatially independent testing data. However, unequal densities of data will still exist, and estimates of model fit will still be more highly influenced by regions of higher density of data. Shown on this map is one possible checker board pattern. After the grid is created, the data used in the analysis to create the model could be the data from the lighter-coloured locations and the data used for cross-validation testing could be from the darker-coloured locations.

Cross-validation techniques with uneven data density

Unfortunately, even a checker board approach to choosing testing data is probably not entirely appropriate for validation models based on checklist data. Figure 6 illustrates why: the uneven distribution of data means that the cross-validation will be most influenced by the areas with highest density of data points, even for species most prevalence in areas with little data. The same problem will occur when assessing the predictive power of a model that describes changes in distribution through the seasons if some seasons will have more available data.

We believe that in general ornithologists would expect that an accurate species distribution model would be one that is accurate across the entire area and time period that is being described. When this is the objective, then we suggest that an appropriate method for selecting testing data is to insure that the testing data are evenly distributed through space and time. For example, one could create a checker board and within each of the “testing” blocks, randomly select an equal number of data to use in the cross validation. Many variants on this general theme could be imagined; Fink et al. (in press) describes the variant currently being used to validate STEM analyses.

We also envision that there are reasons for not wanting to equally weight all parts of a larger region or all time periods when selecting data for cross validation. For example, if you are most interested in producing an accurate model of changing distributions through the period of migration, then it would be appropriate to deliberately insure that a larger proportion of testing data come from this time period. Or, for a species for which you care more about accurate predictions of presence than absence, then it would be appropriate to have a greater proportion of your testing data contain records for which the species was actually recorded. The general point is that cross-validation does not just allow the analyst the flexibility to tailor the selection of data to their specific objectives for model assessment;

instead, cross validation requires that analysts have a clear understanding of their definition of what would make a model accurate.

Discussion

The STEM framework that we have outlined here, and that is described in more detail in Fink et al. (in press) is one possible approach to using checklist data to model species distributions. While we have used decision trees as the base analysis technique that is replicated in each sub-model, the general method does not require the use of decision trees. The base model could be anything else, for example boosted regression trees, linear regression or logistics regression. The STEM framework is a very general method for imposing “just enough” spatial and temporal structure in order to prevent regions of high data density from affecting the model in regions of low data density.

Another possible approach to uneven data density is to pre-process the data to create more equal data densities prior to the analysis. Something akin to this was done by Kéry et al. (in press), because their data were not stored in connection with point locations (in contrast to eBird), but have locations represented as 1 km^2 grid cells. Potential multiple observations were collapsed down to be reporting or non-reporting within a grid cell in each time period. This approach is a specific type of weighting, with lower weights given to each individual datum in a region of higher data density. More sophisticated forms of weighting exist, such as “covariate shifts” developed in the field of machine learning (Quiñonero-Candela *et al*. 2009). One weakness, that we suspect will exist with all weighting methods, is that accuracy of estimates in areas of higher density of data will be reduced, as a trade-off with increasing the relative accuracy of the model in regions of lower data density. A good weighting method will minimize the loss in accuracy in areas of high data density.

With parametric statistical methods to create species distribution models, another approach is available for removing undue influence of data across large distances or long time intervals: the use of spatial or temporal covariance structures as random effects in (generalized) linear mixed models. Adding either a spatial covariance structure (e.g., Lichstein *et al*. 2002) or temporal covariance structure is relatively straightforward. However, for analyses to create models of changes in distribution through both space and time, spatial and temporal covariances (and their interactions) would be needed. In our experience (Fink & Shaby, unpublished data), adding this complex spatio-temporal covariance structure was not analytically tractable for analyses of data from eBird.

While we have presented and are using one specific analyses technique to deal with unequal data density, our STEM method (Fink *et al*. in press), the general message of this paper is that any analysis of data from checklist schemes will need to prevent the results from being overly influenced by spatial regions or times of year from which higher densities of data are available. Both the type of analysis chosen and the method of validation used need to take into account the uneven densities of data likely to come from any checklist scheme.

Acknowledgements

The work described here is part of a larger initiative to gather together and archive all available data on bird distribution and abundance in North America that come from bird counts. We thank the many participants in eBird for contributing the data that we analysed, and the Cornell Lab of Ornithology Information Sciences unit for creating and maintaining the archive of data that we used, especially Kevin Webb, Tom Fredericks, and Tim Levatich. Many people contributed to the development of our understanding of, and approach to species distribution modelling; we especially thank Rich Caruana, Theodoros Damoulas, Giles Hooker, M. Arthur Munson, Mirek Riedewald, Ben Shaby, Daniel Sheldon, David W. Winkler, and Benjamin Zuckerberg for their intellectual contributions. This work was funded by the U.S. National Science Foundation and the Leon Levy Foundation.

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MS received: 15.05.2010
Revised MS accepted: 02.10.2010

Bird Census News 2010, 23/1-2: 21-27

Demographics of the Bonelli’s eagle *Aquila fasciata* population in Cyprus

Nicolaos Kassinis

Abstract. Bonelli’s eagle *Aquila fasciata* is the only eagle species that breeds in Cyprus. Its population is estimated at 31-39 breeding pairs (0.52-0.65 pairs / 100 km²). The SPA network covers 80% of all nest sites. The majority of nests (70%) were built on large Calabrian pine trees *Pinus brutia* with an average altitude of 625m. Nearest neighbor distance (NND) for 27 nesting sites was 7.4 ± 1.1 km (4.1-11.5 km). The mean number of fledglings / successful pair was 1.44 ± 0.53 (1999-2009). Radio telemetry (2002-9) showed that shooting and poisoning are the most significant problems of direct persecution.

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Introduction

The Bonelli’s eagle *Aquila fasciata* is a fairly large-sized eagle typical of Mediterranean landscapes of southern Europe and western Asia. It is considered an important top avian predator in the food-chain of Mediterranean ecosystems (Cheylan 1977, Donazar *et al*. 2005). Since it’s a diurnal raptor representative of Mediterranean ecosystems it can be also regarded as a flagship species for their conservation. This is reinforced by the species unfavourable conservation status (SPEC 3) (BirdLife International 2004) throughout its European range during the last decades. Its European population is estimated at 862-1072 pairs, 65% of which is in the Iberian Peninsula (Arroyo & Ferreiro 2000).

The Bonelli’s eagle is the only eagle species that breeds in Cyprus since the extinction of the Imperial eagle *Aquila heliaca* as a breeding bird in the 1980s (Kourtellarides 1997). It inhabits mountainous terrain, nests mostly at forest edge but usually hunts in more open areas. Immature eagles during dispersal tend to hunt in lowland maquis and agricultural areas where prey is more abundant and more easily obtainable. Also, a few, mostly immature migratory birds appear annually in coastal areas during fall migration. During the late 1950s it was considered common with a population estimate of > 50 pairs (Flint and Steward 1992) however this estimate was based on anecdotal observations. In the 1980s to early 1990s the population declined to less than 20 pairs (BirdLife International / EBCC 2000).

Bonelli’s eagle in Cyprus preys mainly on medium-sized birds and small mammals. Its main prey are Chukar Partridge *Alectoris chukar*, Common Woodpigeon *Columba palumbus,* Rock and feral pigeon *C. livia*, Eurasian Jay *Garrulus glandarius glaszneri*, Corvids (mainly Black-billed Magpie *Pica pica* and Hooded Crow *Corvus corone cornix*), water birds found near wetlands (Coot *Fulica atra* and Little Egret *Egretta garzetta*) and domestic fowl. Game farm chukars released in late summer for augmenting the wild stock for hunting purposes are utilized by eagles that regularly hunt the vicinity of release pens in the countryside.

(Kassinis& Miltiadou 2010). Small mammals, mainly the abundant Black Rat *Rattus rattus* but also the European Hare *Lepus europeus* come second. Reptiles follow in importance, especially large lizards such as the Starred Agama *Laudakia stellio* (occasionally also Schneider's Skink *Eumeces schneiderii*) and to a lesser degree Persian Large Whip Snake *Coluber jugularis*. In 2009, the remains of a Cyprus mouflon lamb *Ovis orientalis ophion* were found for the first time near an eagle's nesting site. The mouflon is the largest mammal on the island and is sympatric to the eagle's stronghold, Pafos forest.

Study area

Cyprus is located at the northeast end of the Mediterranean basin and is the third largest island after Sicily and Sardinia. The island's maximum length and width are roughly 240 km and 100 km, respectively. Neighbouring countries are Turkey 75 km to the north, Syria and Lebanon to the east (105 km and 108 km, respectively), Israel 200 km to the southeast, Egypt 380 km to the south, and Greece 280 km to the west-northwest (the island of Kastellórizo). The island is dominated by two mountain ranges, Troodos Mountains and the smaller Kyrenia Range with the large, central plain of Mesaoria in between. Troodos Mountains cover most of the southern and western parts of the island with their highest peak of Mount Olympus at 1 952 m, located in the centre of the range. Kyrenia Range is narrow, smaller in area and extends along the northern coastline with lower elevations reaching a maximum of 1 024 m (Kyparissovouno peak).

Cyprus is characterized by a typical Mediterranean climate of the extreme type with dry, hot summers, wet, mild winters and short autumn and spring seasons. The island's proximity to the Middle East makes it one of the hottest parts of the Mediterranean. From May to September, temperatures are generally over 30^0C, with cloudless skies and virtually no rain. Spring temperatures average between 23^0C and 27^0C. The winters see milder weather; with January average temperatures ranging from 10^0C on the central plain to 3^0C on the higher parts of Troodos mountains. The average annual precipitation for the year is 462 mm (1971-2000) but it varies from a minimum of 182 mm in 1972/73 and a maximum of 759 mm in 1968/69. Precipitation occurs usually between November and March (80% of the total), with the island averaging 40 days of rainfall every year. Snowfall is frequent in the Troodos range above 1000 m.

Cyprus coverage with high forests (mainly Calabrian pine *Pinus brutia*) reaches 17% of its surface, mainly on the 2 mountain ranges. The endemic Golden oak *Quercus alnifolia* exists in either pure stands or under the conifers in altitude over 700 m across the Troodos range. Dominant shrubs typical of Mediterranean landscapes occur; *Olea / Ceratonia* maquis with lower thorny phrygana are typical at lower ground whereas *Juniperus phoenicea* mattoral is characteristic of the coastal areas (Tsintides *et al.* 2007).

Methodology

This study presents a summary of the species monitoring during the last decade (1999-2009) in the area controlled by the Cyprus government (excluding the Turkish-occupied northern part of Cyprus). In the northern part of the island the species is present in the Kyrenia range, but the situation is obscure; with a possible population of 3 breeding pairs (Whaley & Dawes 2003). In most Mediterranean countries the Bonelli's eagle nests on cliffs.

In Cyprus however, nests are mainly found on large Calabrian Pine trees. Given this fact, all past information on breeding sites was used as much as possible. We collected also data by means of personal inquiries to local people, and by driving along main and secondary roads in potentially suitable habitat. Since the eagle resides in mountainous areas, all mountain ranges were rated as potentially suitable nesting habitat and surveyed. The initial field visits started in mid November when the eagles start displaying and carrying nesting material. Monitoring was carried out in all potential nest sites that were marked on a map (Gilbert et al. 1998). Potential nest sites were mainly steep forest slopes with large Pine trees and relatively scarce vegetation. Nesting trees are usually large in order to support the massive nesting structure. Large cliff formations in remote areas were also searched.

All eagle observations were mapped. The large state forests of Pafos, Troodos, Adelfoi, Macheras and Limassol were searched more intensively due to the availability of more potentially suitable habitat for the species and due to existing knowledge for the species' presence. The number of occupied territories and breeding pairs were recorded. Proof of occupancy of a territory (home range) was established by evidence of territoriality (either seeing 2 eagles together, or witnessing display) or by observing birds carrying nest material. Evidence of a breeding pair was established if copulation, courtship behaviour and finally incubation were witnessed (Hardey *et al*. 2006). When an active nest was spotted (presence of eggs, incubating female or young), regular follow-up checks were carried out (March-April for incubation, May-June for estimating number of fledglings leaving the nest) with a spotting scope from a safe distance in order to record the beginning of the nesting period (incubation, hatching, care of young and fledging). The number of nesting trees in the vicinity of each nest site was recorded. The location of nest sites was recorded with a Garmin GPS receiver (Model GPS map60Cx; Garmin International, Olathe, Kansas, U.S.A.).

Radio tracking information was collected during a parallel eagle tagging project (2002-2009) when eagles were trapped outside the breeding season in areas they were known to frequent in order to hunt. The traps were set with a live bird-bait and eagles of any age group caught were measured, blood sampled, radio-tagged and released. Tags used were backpack transmitters with a mortality /activity sensor provided by BIOTRACK (Wareham, BH20 5AX, UK). The harness used to adjust the tag on the eagle followed Kenward (1987). Radio signals were checked at least once a week usually from fixed, well distributed high observation points.

Results

Breeding population

Thirty-one pairs were located holding breeding territories. Eight more sightings need further investigation to prove territorial occupancy. Twenty six nest sites were located. The current Bonelli's eagle population in Cyprus is estimated at 31-39 breeding pairs. Its stronghold is the Pafos Forest, a large, state-owned area of 62 000 ha with a high eagle density estimated at 2-2.5 breeding pairs per 100 km² with a total nesting population of 12-15 breeding pairs, with most nesting sites located at the forest's edge. The average island-wide Bonelli's eagle density was estimated between 0.52 and 0.65 pairs / 100 km². Nearest neighbour distance (NND) for 27 monitored nesting sites (Figure 1) was 7.4 ± 1.1 km (range min 4.1-max 11.5 km) whereas in Pafos Forest the average distance between neighbouring pairs was smaller (6.2 km)

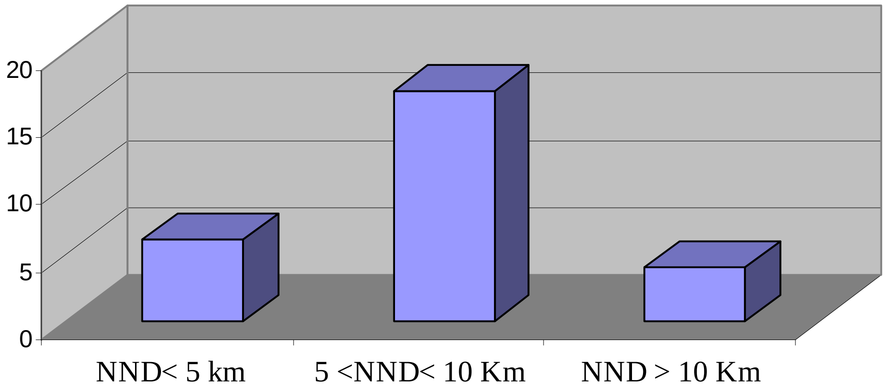

Figure 1. Nearest neighbour distance (NND) for Bonelli's eagle nest sites in Cyprus (N=27).

Productivity

Most pairs start incubation at the end of January – beginning of February. Clutch size is usually 2 and rarely 3 eggs. Young start hatching the last week of February to mid-March and the first eaglets fledge from late April - beginning of May until the beginning of June. Productivity was measured as the number of young fledged per successful pair (Hardey *et al*. 2006). From 1999 to 2009, 89 chicks fledged from 62 successful nests that were closely monitored (average brood size at fledging was 1.44 ± 0.53 fledglings / successful pair). From these, 36 nests (58 %) produced 1 young, 25 nests (40 %) produced 2 young, and 1 nest produced 3 fledged young (Figure 2).

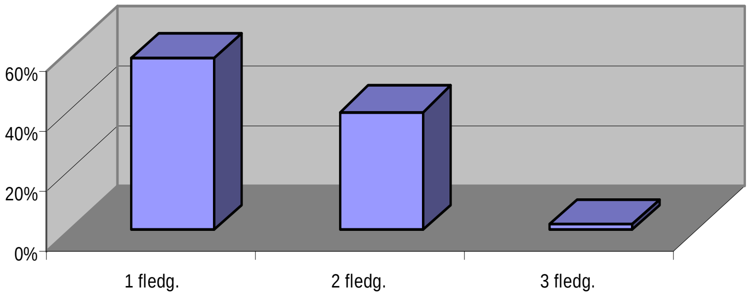

Figure 2. Brood size frequency from 62 successful nests, 1999-2009.

Nest sites

Seventy percent (70%) of the pairs (17 pairs) monitored nested on large Calabrian pine trees (> 15m high), 6 pairs occupied cliff nests (about 24%) whereas 2 pairs had both a tree and a cliff nest. The average number of alternative nesting trees per nesting site was 2.23 ± 0.66 (range 1-3) for 17 exclusive tree-nesters. Nesting cliffs were usually on remote and extensive cliff formations, or deep, high-walled ravines. The average nest site altitude was 625 ± 257m a.s.l (N = 22), even though it varied substantially from a minimum of 55 m up to a maximum of 1200m a.s.l. (Figure 3).

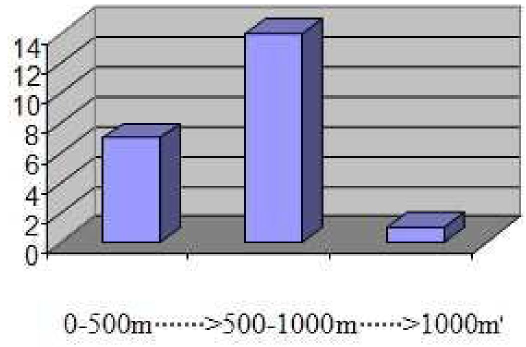

Figure 3. Nest site altitude (mean = 625 ± 257m a.s.l) for Bonelli's eagle in Cyprus (N=22).

Radio telemetry – causes of mortality

From 2002 to 2009 14 eagles including 4 adults, 7 immature and 3 juvenile birds were radio-tagged. Three of the 4 adults caught, tagged and released were part of breeding pairs and were caught 23 km (in January), 4 km (in September) and 1.5 km (in October) from their respective nesting sites.

One eagle caught and tagged as a juvenile in 2005, paired and bred successfully in 2008 and fledged 1 eaglet. This eagle was caught and tagged 38 km away from its recent nesting site of 2008. Another eagle caught and tagged as immature in 2008, paired and bred successfully in 2010 and fledged 1 eaglet. This eagle was caught 46 km away from its current nesting site. During this 8-year of radio-tracking, 4 tagged eagles (28%) died due to direct human persecution: 2 immature birds were shot, 1 juvenile bird was poisoned whereas only the tag was found from the 4th bird (adult). The cause of death was probably human-related.

Discussion

The Bonelli's eagle population in Cyprus is considered stable and in good status. The average brood size at fledgling estimated in this study was similar to studies in Israel in the mid 1970s (Newton 1979). Its population density is higher compared to densities recorded in Spain and south France (Real & Manosa 1997). The species is well distributed across the island, possibly due to the abundance and even distribution of its major prey species such as the Chukar partridge, *Columbidae*, rats and common reptiles (mainly the Starred Agama lizards and Persian Large Whip Snakes). The availability and utilization of such a wide range of locally abundant prey is a major factor contributing to the survival of a raptor such as the Bonelli's eagle (Carrete *et a*l. 2002).

The spatial nest site distribution is probably affected by both the local topography and by the availability of suitable nesting trees away from disturbance. The mean number of alternative nesting trees / nest site is relatively small compared to Ontiveros (1999) wide range of 1-18 nests / nest site. The species weaker preference for cliff nest sites may be attributed to their closer proximity to human habitation and to the extensive road system near cliffs (compared to the more distant forest slopes), but also to the expansion during the last decade of a smaller raptor, the Long-legged Buzzard *Buteo rufinus*, a cliff-nester. In the vicinity of two eagle eyries (< 1.5 km), active Long-legged Buzzard nest sites have been located (Kassinis, 2007).

Radio telemetry data showed that shooting and poisoning are still the most significant problems of direct persecution even though such incidents have declined since the 1980s. Poisoning is a local problem in areas with extensive livestock husbandry where shepherds poison carcasses to decrease lamb/kid depredations by foxes. Moreover, the current development of wind farms in wilderness areas pose a new threat to the species along with other resident raptors such as the endangered Eurasian griffon vulture *Gyps fulvus* but also migratory raptors that pass through the island in large numbers such as *Pernis apivorus*, *Buteo* spp. In addition, unlimited access even in the most remote areas through an extensive network of secondary dirt road, forest tracks and firebreak lanes, causes disturbance to breeding pairs. Housing development in the countryside also increases encroachment, habitat degradation and disturbance.

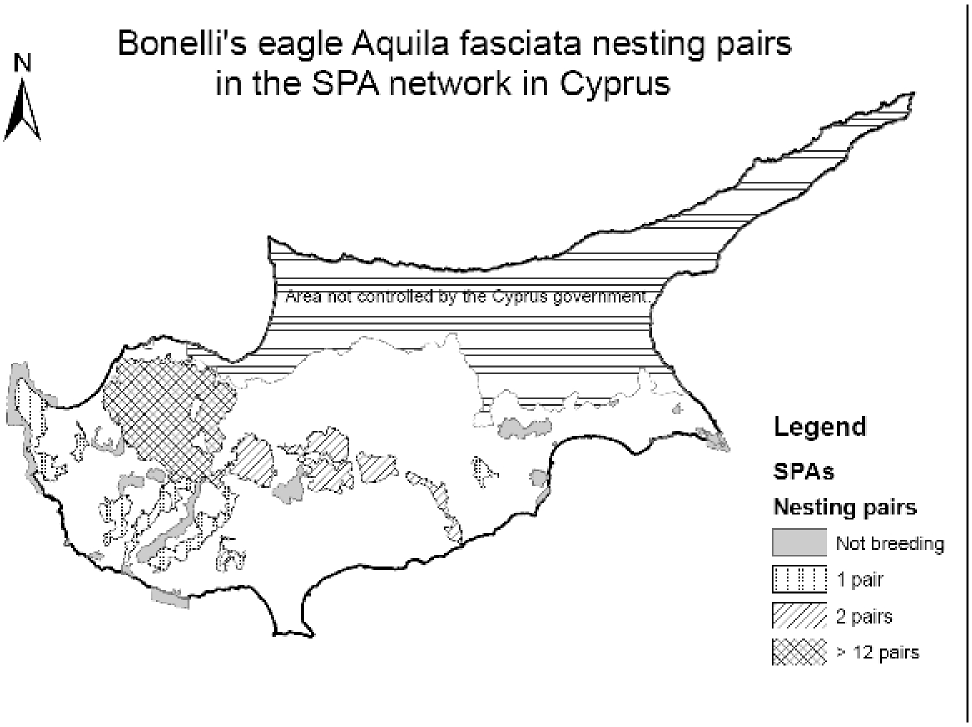

Figure 4. Nesting pairs of Bonelli’s eagle in SPAs in Cyprus

The Bonelli’s eagle is an Annex I species of the EU Bird’s Directive and thus enjoys a strict protection status. This is achieved through the protection of its nest sites in the NATURA 2000 network and the designation of its major breeding areas as Wildlife Conservation Areas where hunting is prohibited according to Cypriot legislation. In Cyprus the total coverage of the 29 Special Protection Areas (SPAs) is 148 313 ha or 16% of the island’s surface (24% of the area controlled by the Cyprus government). The SPA network covers 80% of all nest sites for Bonelli’s eagle including all the high-density eagle areas (Figure 4).

Acknowledgements

Thanks go to P. Hadgigerou, M. Papadopoulos, A. Lysandrou, K. Nicolaou, T. Evstathiou, P. Prastitis, C. Demetriou, I. Kyriakou, G. Pitharides , A. Kreouzos and C.

Hadjistillis for their valuable help with the field work and data collection. Also thanks to P. Anagiotos and P. Panagides for making the distribution map and to I. Charalambidou for reviewing the original manuscript and providing valuable suggestions.

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MS received: 15.05.2010
Revised MS accepted: 25.10.2010

Bird Census News 2010, 23/1-2: 29-40

Use of Species Distribution Modelling based on data from the African Waterbird Census to predict waterbird distributions in Africa and identify gaps in knowledge of distribution

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Abstract. The identification of important sites for waterbirds in Africa is a priority aspect for the African-Eurasian Agreement. Although some areas have been recognized, information about the distribution of several species is still unclear making the process difficult. We modeled the distribution of a target group of waterbird species using Maximum Entropy of Species Distribution Models (MAXENT) and data from the African Waterbird Census. We generated 47 reliable models that were aggregated to identify areas with the highest probability of presence for the target species, overall. Excluding areas already protected by the network of Important Bird Areas and Protected Areas, we identify approximately 3.6% of mainland Africa as sites of high priority to perform gap filling surveys. The methods used in this project may be useful in the future for the identification of target areas for biodiversity conservation in areas where extensive surveys are not possible.

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Introduction

The African-Eurasian Waterbird Agreement requires parties to identify all sites of international or national importance as a matter of priority. Although internationally important sites have been already identified through Wetlands International's African Waterbird Census (Diagana & Dodman 2006) and through BirdLife International's Important Bird Area (Fishpool & Evans 2001) programmes, knowledge concerning the location of key sites for some waterbird species is still incomplete.

The AEWA Implementation Priorities highlight the need for specific gap filling surveys and supporting this objective, the Wings Over Wetlands project organized a series of gap identification workshops, which were based on expert opinion. Recognizing the potential limitations of having based gap identification on limited knowledge of species distribution, the current study aims to complement the results of expert opinion with the results of distribution modeling using information from the African Waterbird Census.

Methods

Study area

Africa, which covers 30 221 532 km^2 is the second largest continent in the world. Approximately 2 313 bird species occur in this vast continent and its associated islands (Fishpool & Evans 2001) and around 218 of those species are considered waterbirds (Wetlands International 2006, Diagana & Dodman 2006). While land bird richness is mainly located in the tropical areas of Africa, waterbird species richness is higher in the eastern and southern parts of the continent (Guillet & Crowe 1985).

Some regions in Africa are of high importance as wintering areas for Palearctic migratory waterbirds. It is estimated that around 5000 million individuals of these species spend their winter in sub-Saharan Africa (Newton 2008) and the survival of some, e.g. *Gallinago media*, *Charadrius leschenaultia*, *Numenius phaeopus, N. arquata, Calidris alba, C. ferruginea,* depends on the availability and quality of these wintering grounds (Delany *et al*. 2009).

Species selection

In order to assist gap identification, this study has focused only on congregatory waterbird species (species that concentrate at least 1% of their population in a particular area; this criteria is widely used to support the conservation of sites, Wetlands International, 2006) currently poorly covered by the Important Bird Area network in Africa (Nagy *et al*. 2008). In total 59 waterbird species matched these criteria and are included in the analysis (Table 1).

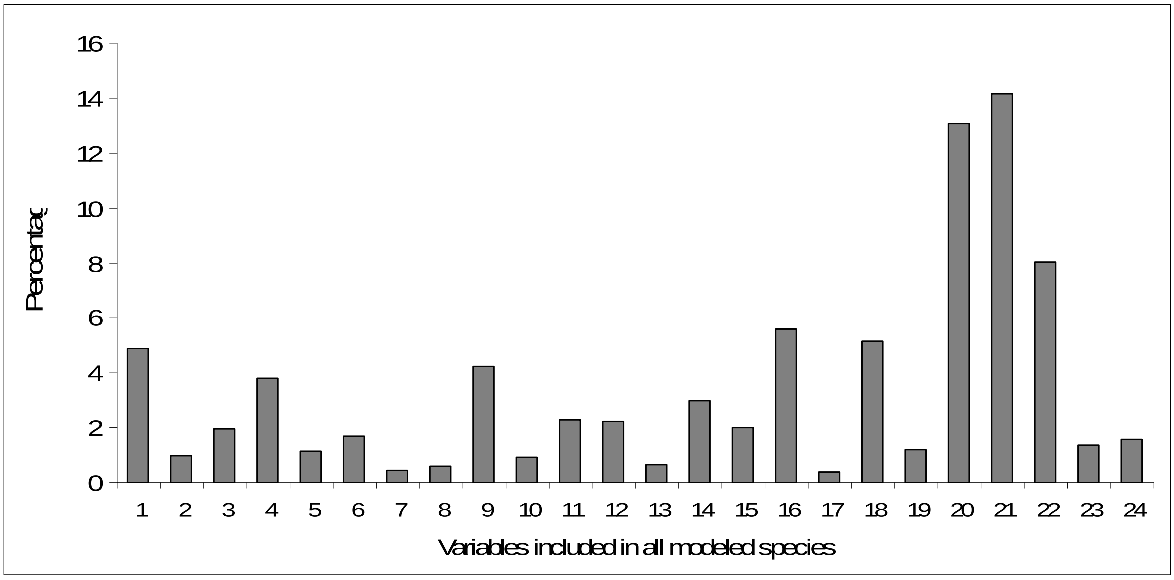
Species	**Category (BirdLife International)**	**N of sites (Training)**	**N of sites (Test)**	**Total N of sites**	**AUC Training**	**AUC Test**
Podiceps cristatus infuscatu	**LC**	**79**	**33**	**112**	**0.995**	**0.975**
Podiceps nigricollis nigricollis	**LC**	**73**	**30**	**103**	**0.984**	**0.975**
Tachybaptus pelzelnii	VU	9	0	9	0.994	
Pelecanus onocrotalus	**LC**	**97**	**41**	**138**	**0.967**	**0.953**
Pelecanus rufescens	**LC**	**60**	**25**	**85**	**0.977**	**0.945**
Phalacrocorax carbo lucidus	**LC**	**152**	**65**	**217**	**0.983**	**0.969**
Phalacrocorax africanu		**343**	**146**	**489**	**0.962**	**0.945**
Anhinga rufa	**LC**	**172**	**73**	**245**	**0.975**	**0.934**
Egretta vinaceigula	VU	12	1	13	0.995	1.000
Ardea humbloti	EN	10	2	12	0.999	0.997
Ardea purpurea	**LC**	**238**	**101**	**339**	**0.960**	**0.942**
Mesophoyx intermedia	**LC**	**202**	**86**	**288**	**0.968**	**0.939**
Ardeola idae	EN	12	2	14	0.988	0.867
Ciconia abdimii	**LC**	**38**	**16**	**54**	**0.950**	**0.873**
Plegadis falcinellus falcinellus	**LC**	**239**	**102**	**341**	**0.972**	**0.951**
Platalea alba	**LC**	**213**	**90**	**303**	**0.977**	**0.958**
Phoenicopterus minor	**NT**	**82**	**35**	**117**	**0.983**	**0.941**
Dendrocygna bicolor	**LC**	**144**	**61**	**205**	**0.974**	**0.927**
Dendrocygna viduata	**LC**	**285**	**122**	**407**	**0.964**	**0.935**
Thalassornis leuconotus	**LC**	**73**	**30**	**103**	**0.973**	**0.957**
Oxyura maccoa	**NT**	**71**	**30**	**101**	**0.990**	**0.978**
Tadorna cana	**LC**	**82**	**35**	**117**	**0.996**	**0.960**
Plectropterus gambensis	**LC**	**252**	**107**	**359**	**0.966**	**0.940**
Sarkidiornis melanotos	**LC**	**187**	**79**	**266**	**0.962**	**0.941**
Anas hottentota	**LC**	**117**	**49**	**166**	**0.981**	**0.943**
Anas querquedula	**LC**	**117**	**50**	**167**	**0.991**	**0.973**
Anas smithii	**LC**	**121**	**51**	**172**	**0.995**	**0.981**

Nettapus auritus	**LC**	**73**	**30**	**103**	**0.960**	**0.940**
Balearica pavonina	**NT**	**63**	**27**	**90**	**0.991**	**0.958**
Grus paradisea	VU	8	0	8	0.950	
Grus carunculatus	VU	15	3	18	0.970	0.994
Gallinula angulata	**LC**	**37**	**15**	**52**	**0.978**	**0.917**
Fulica cristata	**LC**	**168**	**72**	**240**	**0.987**	**0.963**
Haematopus moquini	**NT**	**24**	**9**	**33**	**0.997**	**0.976**
Himantopus himantopus	**LC**	**372**	**159**	**531**	**0.963**	**0.938**
Recurvirostra avosetta	**LC**	**148**	**63**	**211**	**0.974**	**0.954**
Glareola pratincola	**LC**	**96**	**40**	**136**	**0.982**	**0.961**
Glareola nordmanni	NT	18	4	22	0.992	0.750
Charadrius pecuarius	**LC**	**193**	**82**	**275**	**0.973**	**0.953**
Charadrius pallidus	**NT**	**26**	**11**	**37**	**0.994**	**0.985**
Charadrius alexandrinus	**LC**	**35**	**15**	**50**	**0.996**	**0.951**
Charadrius leschenaultii	LC	10	4	14	0.950	0.828
Vanellus melanopterus	LC	8	3	11	0.899	0.995
Gallinago media	**NT**	**64**	**27**	**91**	**0.977**	**0.933**
Limosa limosa	**NT**	**107**	**45**	**152**	**0.991**	**0.967**
Numenius phaeopus	**LC**	**61**	**25**	**86**	**0.994**	**0.977**
Numenius arquata	**NT**	**65**	**26**	**91**	**0.993**	**0.985**
Arenaria interpres	**LC**	**53**	**22**	**75**	**0.989**	**0.993**
Calidris alba	**LC**	**49**	**20**	**69**	**0.990**	**0.965**
Calidris ferruginea	**LC**	**161**	**69**	**230**	**0.978**	**0.969**
Larus dominicanus	**LC**	**34**	**14**	**48**	**0.996**	**1.000**
Larus cirrocephalus	**LC**	**157**	**67**	**224**	**0.981**	**0.949**
Sterna caspia	**LC**	**98**	**42**	**140**	**0.997**	**0.961**
Sterna maxima	**LC**	**28**	**11**	**39**	**0.998**	**1.000**
Sterna bengalensis	LC	18	7	25	0.998	0.998
Sterna albifrons	**LC**	**53**	**22**	**75**	**0.989**	**0.996**
Sterna balaenarum	NT	6	2	8	0.997	0.989
Chlidonias hybridus	**LC**	**161**	**69**	**230**	**0.977**	**0.948**
Rynchops flavirostris	NT	20	4	24	0.975	0.990

Table 1. Species whose distribution were modeled in the study, the sample sizes for training and test data, and their respective AUC values. The 47 species selected to identify the relevant areas are in bold.

Data

We used the African Waterbird Census (AfWC) Database of Wetlands International to obtain the localities where each species had been recorded. The dataset contains 197 043 records from waterbird counts in Africa, from 1 517 sites located in 41 sub-Saharan African countries. The information used from the database corresponds to observations collected mainly from 1997 to 2007.



1. Annual Mean Temperature
2. Mean Diurnal Range
3. Isothermality
4. Temperature Seasonality
5. Max T Warmest Month
6. Min T of Coldest Month
7. T Annual Range
8. Mean T of Wettest Quarter
9. Mean T of Driest Quarter
10. Mean T of Warmest Quarter
11. Mean T of Coldest Quarter
12. Annual Precipitation
13. Precipitation Wettest Month
14. Precipitation Driest Month
15. Precipitation Seasonality
16. Precipitation Wettest Quarter
17. Precipitation Driest Quarter
18. Precipitation Warmest Quarter
19. Precipitation Coldest Quarter
20. Altitude
21. Wetlands (WWF Wetlands Layer)
22. Herbaceous Cover (Global Land Cover)
23. Tree cover (Global Land Cover Facility)
24. Bare cover (Global Land Cover Facility)

Figure 1. Average percentage of contribution for each variable calculated from the jacknife procedure from MAXENT for all modelled species. Each variable is denoted as a number in the graph and the complete name shown below. Unless otherwise noted, the source is Worldclim (Hijmans R. J. et al. 2005). Other sources were the Global Land Cover Facility (www.landcover.org) and the Wetlands layer (Lehner, B. and P. Döll 2004).

Selection of environmental variables

The modeling of species distribution at large geographic scales is based on the principle of finding the relationship between various environmental factors and the occurrence of the species. In large scale studies, the availability and quality of data layers inevitably influences the selection of environmental variables.

Considering the requirements of waterbirds in general, 24 environmental variables were obtained from three different sources: (i) Worldclim -19 climatic variables plus altitude - (Hijmans *et al*. 2005), (ii) the Global Land Cover Facility -three habitat variables- (Hansen *et al.* 2002), and (iii) the Lakes and Wetlands Database information about aquatic ecosystems (Lehner & Döll 2004) (see Figure 1). The 24 environmental variables were processed using ArcGis 9.3 (ESRI) and DivaGis, and their cell sizes were standardized to a 1 km^2 resolution.

Modelling

The distribution of each selected species was modeled using MAXENT version 3.2.19 (Phillips *et al*. 2006) with the localities obtained from the AfWC as dependent and the 24 environmental variables as predictor variables. For each model we obtained response and rediction curves to assess the performance of the models and run the in-built jackknife procedures to assess the estimating value of each variable. We produced logistic probability maps of species presence (Phillips & Dudik 2008). To avoid over-training of the model at localities with multiple records for the same species, duplicates were removed. The number of background points was set at the default value (10 000), since it has been shown that the model's performance increases when the numbers of background points is up or around this value (Phillips & Dudik 2008). Finally, 30% of the records were randomly selected as test data to evaluate the model's predictive power on independent data, and the remaining 70% were used to train the model. However, the percentage of test data was reduced when the number of records was limited as in the case of *Egretta vinaceigula, Ardeola idea, Grus paradisea, Grus carunculatus, Glareola nordmanii* and *Vanellus gregarious* .

In addition to the evaluation of the model through splitting the dataset, model performance for four species: *Phoenicopterus minor, Ardeola idea, Gallinago media* and *Glareola normanii* was assessed using completely independent datasets of localities from their respective action plans (Childress *et al*. 2007, Kariuki N. & Sande E. 2008*,* Kålås, J.A. 2004*,* Belik & Lebedeva 2004). This data was included as a test sample file, and a model for each of these species was rerun.

Identification of priority areas for gap filling surveys

The species distribution models were converted from ASCII to grid files using ArcGis 9.3. (ESRI) and priority areas were identified by aggregating the probabilities of occurrence of individual species. Before aggregation, two processes were performed. First, areas below the threshold of maximum training sensitivity plus specificity, calculated directly by MAXENT, were excluded from this analysis assuming that probability below these thresholds represent absence. This threshold is recommended over other approaches in an evaluation of the performance of twelve different methods for setting presence/absence areas (Liu et al. 2005). Second, following the suggestion of Wisz *et al*. (2008), only species with more than 30 localities as sample size were included. As a result 47 species were incorporated in the final sum of probabilities map (Table 1).

The aggregated occurrence probabilities were classified in ten quantiles and locations in the two highest quantiles were considered the most important areas for gap filling surveys, based on the aggregated probability of finding one or more of the target species.

In order to focus gap filling surveys on areas not yet well covered, we created a map of already known hot spots for waterbirds. We obtained polygons of protected areas from the World Database on Protected Areas (WDPA; World Database on Protected Areas 2009) and from the Important Bird Areas (IBAs) from BirdLife International (BirdLife International 2010a). The two layers were merged and a five kilometer buffer was created around all protected areas and IBAs. Then, by subtracting the latter layer from the probability of occurrence layer we produced a map highlighting target areas for gap filling surveys for waterbirds in Africa.

Results

Model accuracy

According to the Area Under the Curve values (AUC) all models are considered to have a high quality of predictions in test and training data (Elith *et al*. 2006, Wisz *et al*. 2008). In total 98% of the training AUC values were above 0.95, close to a perfect discrimination (AUC value=1). Similarly, 94% of the 59 models had AUC value above 0.90 for the test data. However, models generated with lower number of samples (<30) over-predicted the species' distribution. This was evident when the modeled distributions were overlapped with the polygons of the BirdLife's distribution ranges (BirdLife International 2010b). This was particularly evident for *Grus paradisea*, *Ardeola idae* and *Tachybaptus pelzelnii*. Models generated based on sample size higher than 30, showed a better prediction, and almost the entire area of predicted presence was located within its geographic distribution. Examples of results are shown in Figures 2 and 3 for *Sarkidiornis melanotos*, *Phalacrocorax africanus*, *Dendrocygna bicolor* and *Ciconia abdimii*. Generally, models based on 30 or more occupied sites resulted in finer resolution maps than the distribution maps drawn on the basis of a literature review by BirdLife International (in Nagy *et al*. 2008).

The evaluation of models using completely independent datasets showed high model performance for *Phoenicopterus minor* (AUC training 0.958, AUC test = 0.953). . This prediction was similar to the one obtained in the test using 30% of the sample. However, the three other species evaluated with independent datasets had lower AUC values in tests: *Ardeola idea* (0.794), *Gallinago media* (0.794) and *Glareola normanii* (0.804)

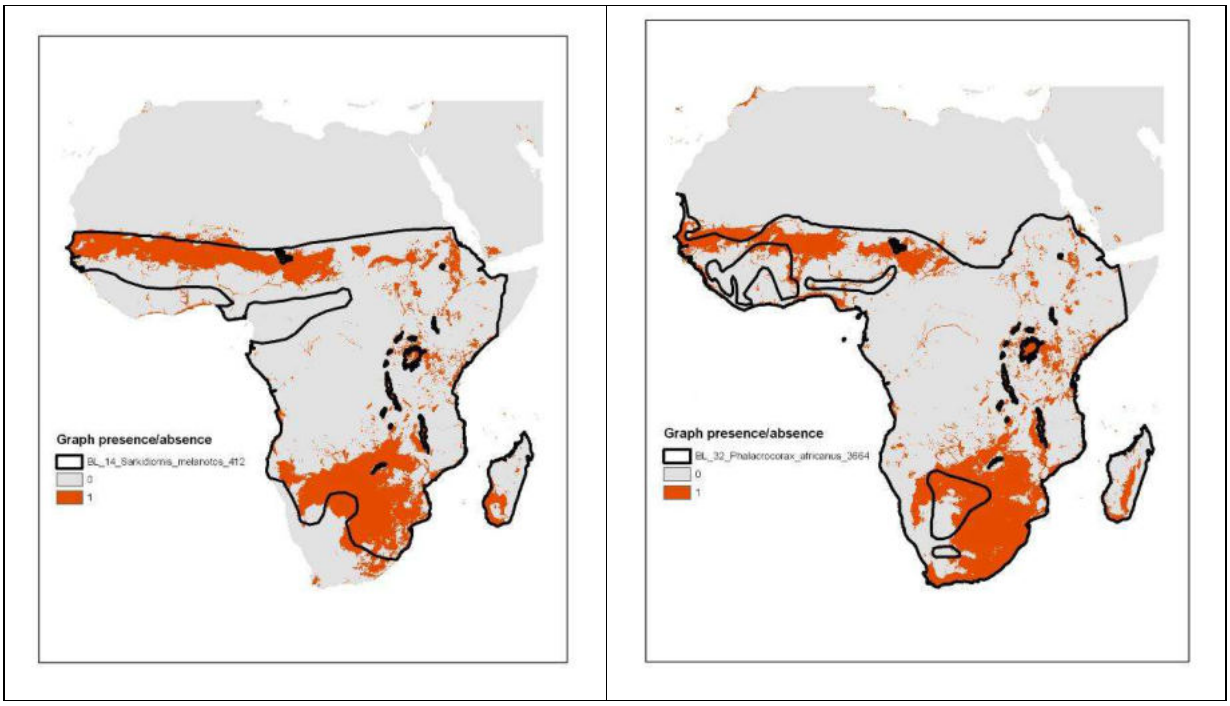


Figure 2. On the left the map of presence (1-dark) or absence (0-grey) for *Sarkidiornis melanotos* (266 records) and the distribution from the WOW project. On the right, the same information is presented for *Phalacrocorax africanus* (489 records).

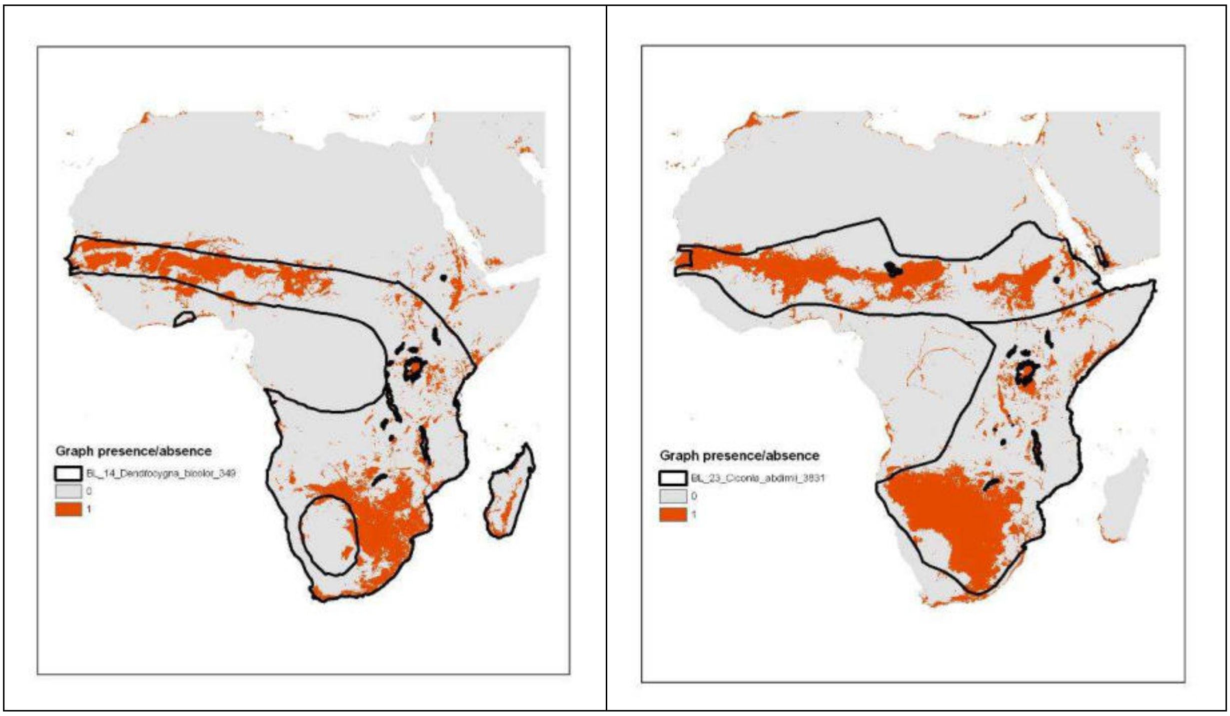


Figure 3. On the left, the map of presence (1-dark) or absence (0-grey) for *Dendrocygna bicolor* (205 records) and the distribution from the WOW project. On the right, the same information is presented for *Ciconia abdimii* (54 records).

Importance of environmental variables

On average, the variables with highest contributions to the model predictions were, in order: area of wetlands, altitude, herbaceous cover, precipitation during the wettest quarter, mean temperature during the driest quarter, annual mean temperature, and temperature seasonality. The remaining variables had a low contribution to the building of the models (Figure 1).

Priority areas for gap filling surveys

The high priority areas identified on the aggregated probability map covered 1 506 578 km^2, located mainly in sub-Saharan Africa, eastern Africa including The Rift Valley, South Africa and the western coast of Madagascar (Figure 4). This represents approximately 3.6% of the land surface of Africa. Of this, protected areas and IBAs cover 484 520 km^2. Consequently, we identified an additional 1 022 058 km^2 to be surveyed (Figure 5).

Analysis

According to Swet's scale of the Area Under the Curve values, all models show high prediction accuracy (Elith *et al*. 2006). On average, the AUCs obtained by training and test data indicate models with high discriminatory ability between areas of presence and absence (0.976 training 0.921 for test data).

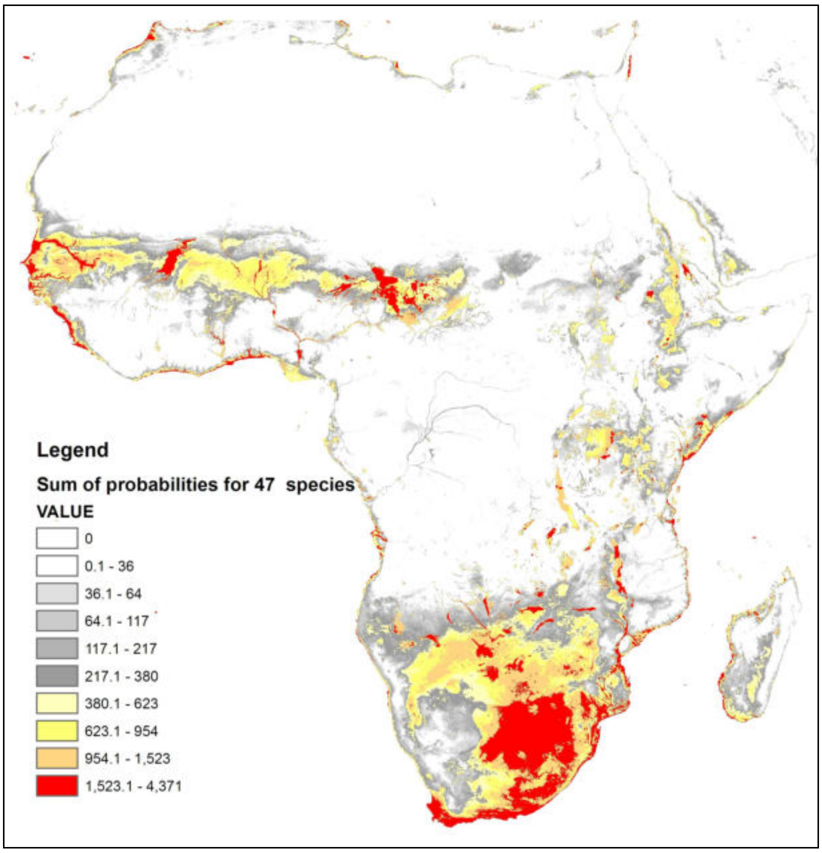


Figure 4. The figure shows in dark the areas where the sum of percentage probabilities from all 47 species had highest values. The range of probabilities was categorized in quantiles.

Models with these characteristics reduced the commission error in predictions of species occurrence in areas outside its distribution range, and are suitable to build maps of species richness for conservation purposes (Pineda & Lobo 2009). However, species with a lower number of samples tend to show an over-prediction in the distribution. This trend was evident in the evaluation of 28 species for which the percentage of area predicted outside their BirdLife's species distribution maps (BirdLife 2010b) was compared against the area predicted inside their distribution ranges ($p<0.001$, Figure 6). Nevertheless, the influence of sample size accounted for 38% of the variability on the commission error obtained. This information partially supports the conservative approach of Wisz et al (2008) to consider models with lower sample size (<30) as experimental, and our decision to exclude models with a low number of samples (<30).

The distribution of waterbirds on wetlands depends on a combination of factors including geomorphology, hydrology and climatic aspects (Guillet & Crowe 1995). According to Newton (2008), ecological conditions in Africa depend more on rainfall rather than temperature. In addition, food availability is one of the limiting factors determining the density of waterbird species (Nummi *et al*. 1994). As a consequence, the number of variables included in this study is far from resembling the area where the 'realized niche' for the species' modeled occurs. However, considering the variables included and the order of importance produced by the models, results are in agreement with some of the most important factors that determine the distribution of waterbirds species found in the literature.

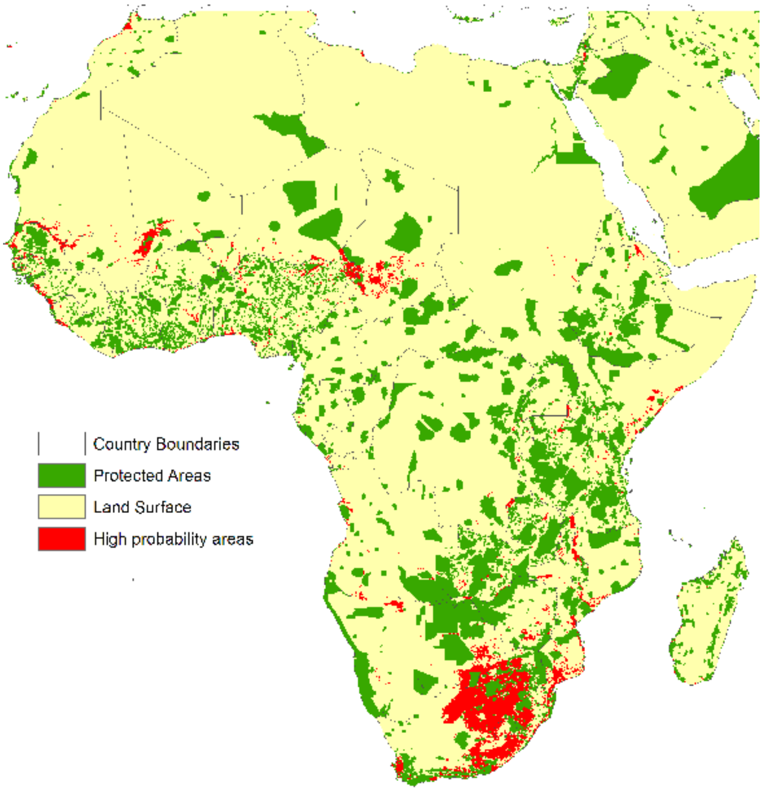


Figure 5. Darkest areas are those selected to perform gap filling surveys. Slightly lighter areas are protected areas.

Not surprisingly, the most important variable in the model generation was the wetlands layer. Herbaceous cover, the second most important variable, may be related to wetland characteristics favoring the presence of some waterbird species in specific areas, as has been shown in other studies (Guillet & Crowe 1995). The other four climatic variables with higher than average percentage contribution to the models are related to evapotranspiration, and consequently water availability. Those were: precipitation during the wettest quarter, mean temperature during the driest quarter, annual mean temperature and temperature seasonality. These are considered determinant factors in defining the presence of wetlands ecosystems (Guillet & Crowe 1985) and consequently waterbirds (Diagana & Dodman 2006).

Previous studies have identified important areas for biodiversity conservation both worldwide and with an Africa focus (Myers *et al*. 2000, Brooks *et al*. 2004, Brooks *et al*. 2001, Rodriguez *et al*. 2004). Among them, this study relates most with the Brooks *et al*. (2001) study, in which a set of priority conservation areas in Africa was defined based on a group of vertebrates, including birds. The top five regions identified by these authors were located in the Eastern Arc Forest, the forest of Upper and Lower Guinea, the Cape Fynbos, the Albertine Rift, and the Ethiopian Highland (Brooks *et al*. 2001). However, none of these studies used the resolution and the specific taxonomic group that were used for the present study. The present project also differs in its purpose of identifying gap of knowledge areas, which may be considered less ambitious than a formal gap analysis (Scott *et al*. 1993).

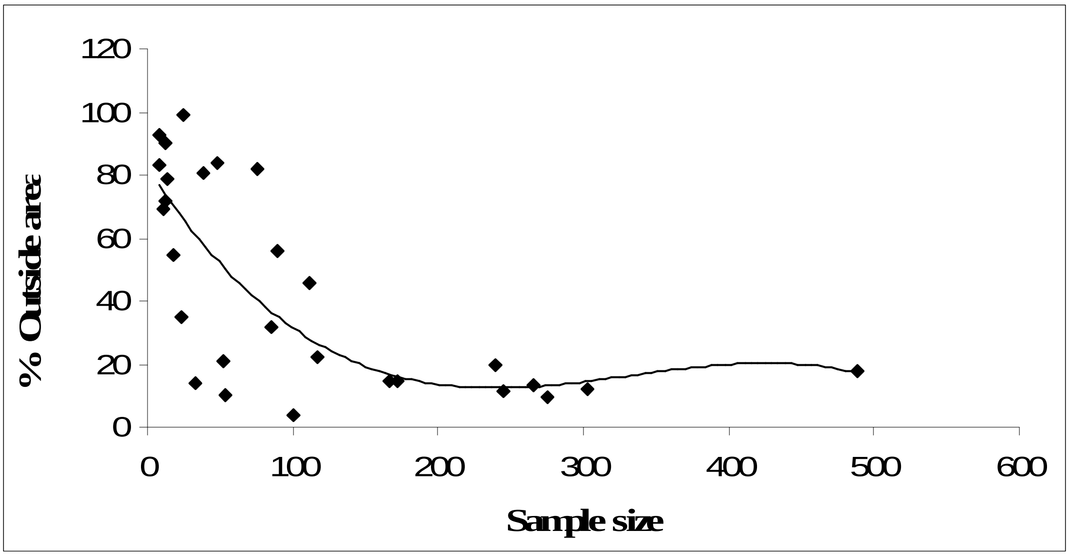


Figure 6. Graph of the percentage of the distribution predicted outside of the distribution area for 28 species, and their respective sample sizes. p<0.001, r=0.38

The location of target areas to perform gap filling surveys showed an overlap with the areas estimated to have high waterbird richness in Africa according to Guillet and Crowe (1985). It is important to clarify that Guillet and Crowe (1985) did not include the northern part of Africa, and Madagascar. The areas identified by this study and by Guillet and Crowe (1985) are: South Africa region, the inland wetlands area, and the mountains of Kenya, Tanzania, Uganda and Mali with an occurrence of at least 85 waterbird species. Other areas with high waterbird richness coincide with locations in western Africa, Mozambique, and coastal areas of Angola and Namibia (Figure 4). This level of overlap implies that the regions found to be important to perform the gap filling surveys would fulfill the knowledge gap in areas where important conservation efforts should be made, due to their waterbird species richness.

Acknowledgments

I am really thankful to my supervisor dr. Joost Duivenvoorden at the University of Amsterdam. His orientation and support were always of essential guidance during this process. I always had important support from Niky Petkov and Simon Delany at Wetlands International. Diego Valbuena provided important input in several aspects related to Geographical Information Systems. BirdLife International kindly provided the information on IBAs and species ranges.

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MS received: 15.06.2010
Revised MS accepted: 30.03.2011

Bird Census News 2010, 23/1-2: 41-47

The distribution of breeding birds in Switzerland in the 1950s compared to the present situation

Peter Knaus

Abstract. A project was launched to document the distribution of breeding birds in the 1950s in Switzerland. This historical atlas allows direct comparisons with the two atlases 1972–1976 and 1993–1996. As many original records as possible at a national level were compiled from notebooks and different archives as well as via standardised interviews with 56 field ornithologists of the older generation. The country was divided into 467 atlas squares (10 x 10 km), but convincing comparisons of the distribution in 1950–1959 with the two published atlases are in general restricted to the Swiss Plateau. Our data document that in the 1950s several farmland and wetland species were widespread across the whole Plateau, but since then lost a considerable part of their range or disappeared completely. On the other hand, we also report on range expansions for some species.

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Introduction

During the 20th century natural habitats came under increasing pressure from changes in land use, habitat destruction, use of pesticides, pollution and human interference. In Switzerland, over 90 % of the wetlands and moorlands have been drained since 1850 and numerous rivers were regulated and canalised to avoid floods and to gain cultivated land. More than 10 million fruit trees were cut since 1950, and in some regions of the country a third of the hedges were destroyed in the 1970s and 1980s (Maumary *et al*. 2007). This led to drastic changes in the avifauna.

A good knowledge of the historical distribution and abundance is an important basis for assessing the current status of bird communities in a region. In Switzerland, the trends of distribution and numbers are documented only since the 1970s, for many species only since around 1990. For the distribution of breeding birds we use the two breeding bird atlases compiled from 1972 to 1976 and from 1993 to 1996, respectively (Schifferli *et al*. 1980, Schmid *et al*. 1998). Data on population trends are available since the mid 1980s (Schmid *et al*. 2001). However many species had suffered massive declines already in the 1970s and before, as intensification of land use started in the 1950s or even earlier.

The project «Avifauna 1950», launched by the Swiss Ornithological Institute in 2007, aims at documenting the distribution of the breeding birds in the 1950s. There is much, mostly anecdotal evidence of the local distribution of bird species of that time period, but so far no effort has been undertaken to compile these mainly original records at a national level. The only nationwide publication covering this period, the book on the breeding birds of Switzerland published in 1962 (Glutz von Blotzheim 1962), gives relatively detailed information on the distribution of many species around the 1950s, but only a few maps are shown. The aim of the project is a breeding bird atlas for the period 1950–1959, allowing direct comparisons with the two atlases 1972–1976 and 1993–1996. The project also aims at raising awareness among politicians, government agencies and the public.

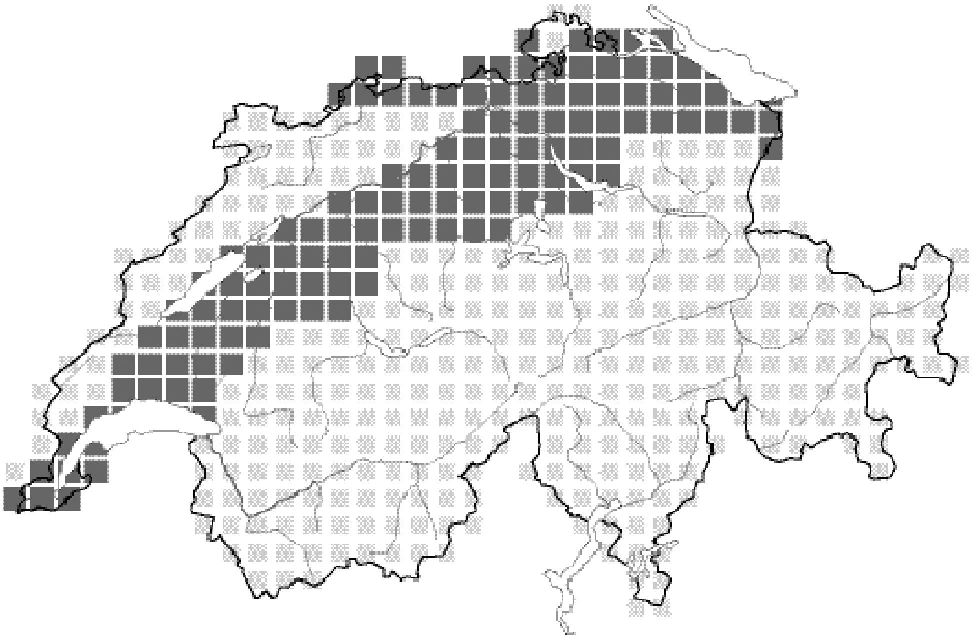

Figure 1. The Plateau is the lowland region between the Jura and the Alps. Atlas squares (10 x 10 km) on the Plateau are dark grey, those outside the Plateau are grey; they are shown with smaller squares on the change maps to reflect less complete coverage.

Methods

Many field ornithologists of the older generation still have a good knowledge of the situation of breeding birds in the 1950s. This knowledge will be lost soon, since most of these observers are aged. Using information in their notebooks and with standardised interviews, this knowledge was used as far as possible. Altogether, 56 interviews were carried out with old-age regional ornithologists by running through a list of the breeding bird species of Switzerland, discussing if the respective species had been present during the breeding season at the time or not. If the species was present, the breeding probability was noted (possible or certain breeding) and, if available, additional information on the abundance, the regularity of breeding and further details. For rarer breeding birds the exact locations were recorded on a large-scale map if possible.

Further sources were the archives of the Swiss Ornithological Institute, such as the archive of the 1962 book, observation data, nest cards and ringing data. Additional information came from regional overviews, publications and private compilations as well as from the specimen lists of the natural history museums of Basle, Berne, Geneva, Fribourg and St. Gallen.

The species were classified as follows:

1. 45 well documented, very rare or rare breeding birds, for which the data were already completely available;
2. 104 scarce or formerly widespread species, for which data were collected as completely as possible;
3. 64 rather common breeding birds with no indications of massive changes in distribution, for which the records were not systematically collected.

To reduce the effort, in general only one observation per atlas square (10 x 10 km) and species was recorded. Similar to the bird atlas 1993–1996 breeding probability (possible, probable and certain breeding) will not be differentiated on the maps. Instead criteria were defined for each species for records to be included on the maps. The criteria list is in general identical to that used for the atlas 1993–1996.

As in the two existing breeding bird atlases, the country was divided in 467 atlas squares (10 x 10 km). Mainly because of the topographic characteristics of Switzerland, the coverage in the 1950s is non-homogeneous. Therefore convincing comparisons of the distribution in 1950–1959 with the two atlases are in general restricted to the Swiss Plateau (in German «Mittelland»), the lowland region between the Jura Mountains and the Swiss Alps, which covers about 30 % of the surface of Switzerland. The biggest changes in land use occurred in the heavily populated Plateau. Atlas squares outside the Plateau are shown by smaller symbols on the change maps, to reflect less complete coverage (Figure 1).

Results

There are enough data for about half of the 213 species to produce representative distribution maps for the 1950s. The data document that in the 1950s several farmland and wetland species were widespread across the whole Plateau: Little Bittern *Ixobrychus minutus*, Common Snipe *Gallinago gallinago*, Eurasian Curlew *Numenius arquata*, Little Owl *Athene noctua*, Eurasian Hoopoe *Upupa epops*, Woodlark *Lullula arborea*, Great Reed Warbler *Acrocephalus arundinaceus*, Icterine Warbler *Hippolais icterina*, Lesser Grey Shrike *Lanius minor*, Great Grey Shrike *L. excubitor*, Woodchat Shrike *L. senator* and Ortolan Bunting *Emberiza hortulana* (Table 1). These species showed marked declines already on the maps 1972–1976 and two of them (Lesser and Great Grey Shrike) had disappeared completely by 1993–1996. For instance, the Eurasian Hoopoe (Figure 2) was still widespread across the whole Plateau in the 1950s. Already until the 1970s, the species disappeared from a part of that region, especially in areas with expanding urban areas. By 1993–1996 the Hoopoe was just an irregular breeder on the Plateau. The reasons for this trend are intensification of land use, agricultural mechanisation, land consolidation and pesticides.

Other breeding birds such as the Grey Partridge *Perdix perdix* and the Corn Crake *Crex crex* were distributed only fragmentarily already in the 1950s (Table 1). Their decline must have started earlier, as at the beginning of the 20th century these species were described as widespread and locally common breeders on the Plateau. In the 1950s, the Corn Crake was found mainly on the Plateau, although with many gaps (Figure 3). However considering the known population fluctuations of this species, the period of 10 years probably gives a rather optimistic image of the distribution. In the 1990s, the distribution was limited even more and breeding attempts occurred almost exclusively in the Jura and the Alps.

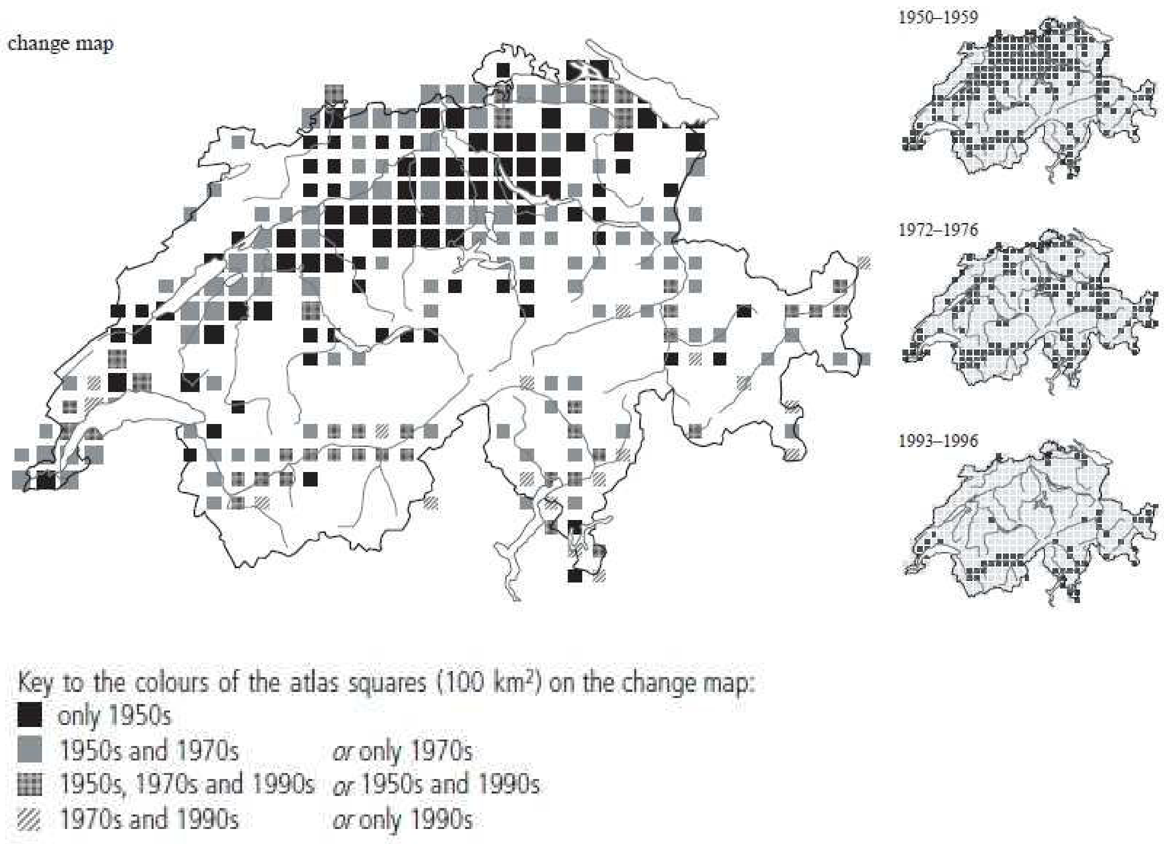

Figure 2. The three distribution maps 1950–1959, 1972–1976 and 1993–1996 (black: at least one record, grey: no record) and the change map of the Eurasian Hoopoe *Upupa epops*

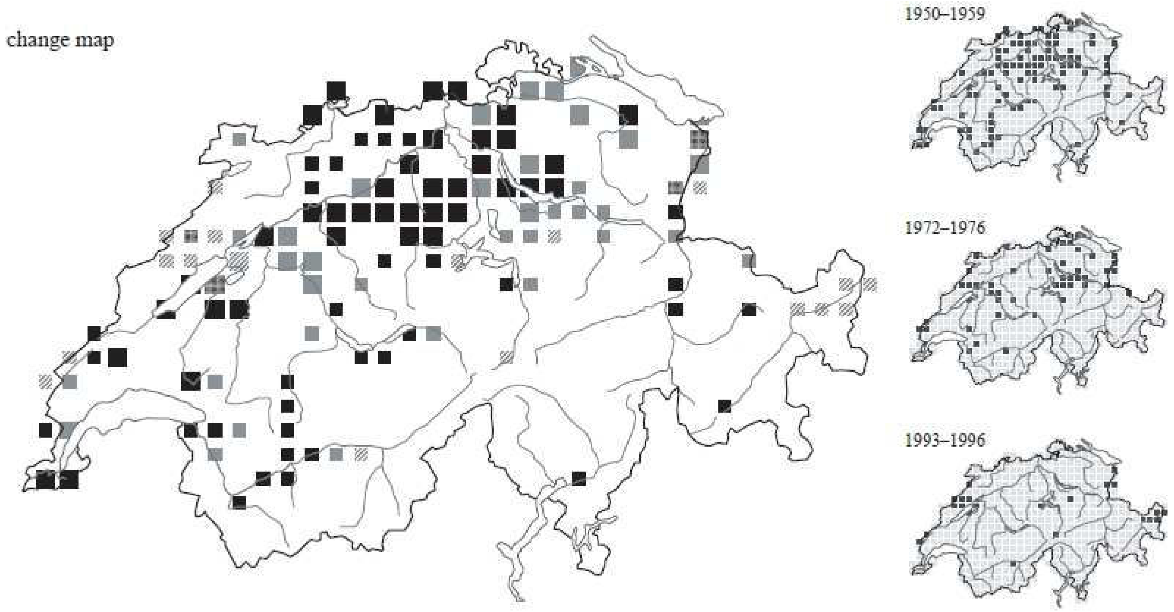

Figure 3. The three distribution maps 1950–1959, 1972–1976 and 1993–1996 and the change map of the Corn Crake *Crex crex* (for square legend see Figure 2).

On the opposite, increases are documented, e.g. for White Stork *Ciconia ciconia*, Black Kite *Milvus migrans*, Red Kite *M. milvus*, Eurasian Collared Dove *Streptopelia decaocto*, Eurasian Stonechat *Saxicola torquatus*, Fieldfare *Turdus pilaris* and Northern Raven *Corvus corax*. For instance, the spread of the Red Kite (Figure 4) can be well shown: In the 1950s, the species was limited mostly to the northern third of Switzerland. By 1993–1996 it had extended the breeding range towards the southwest, south and east and had reached the foothills of the Alps. And this trend still continues today.

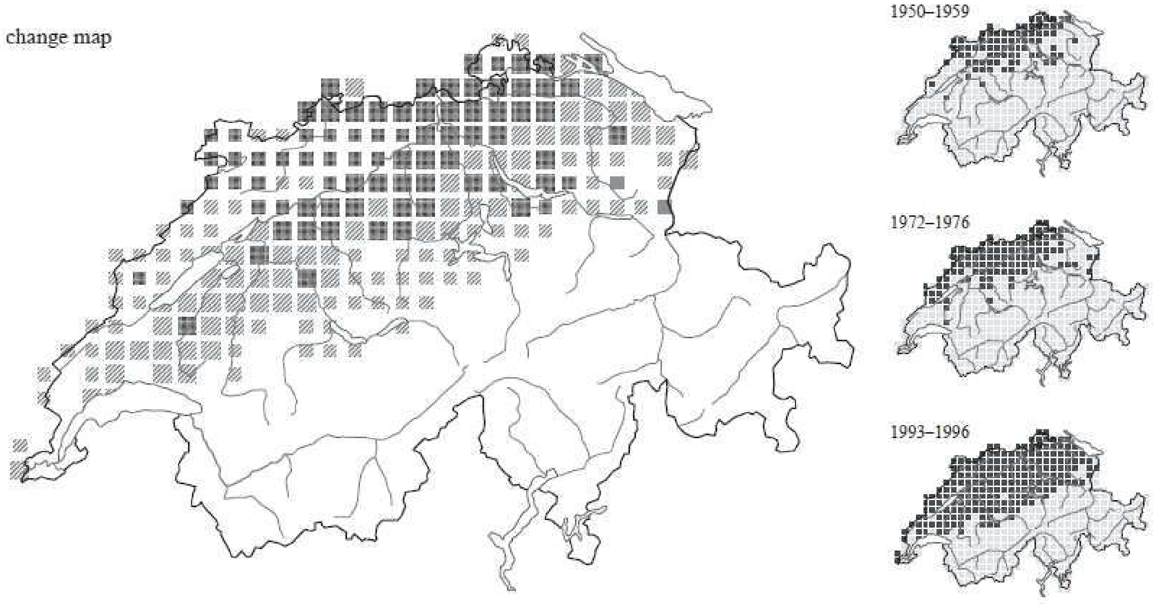

Figure 4. The three distribution maps 1950–1959, 1972–1976 and 1993–1996 and the change map of the Red Kite *Milvus milvus* (for square legend see Figure 2)

Discussion

These results help to better understand today's distribution of the breeding birds of Switzerland. Many species lost a considerable part of their range on the Plateau already between the 1950s and the 1970s. Some wetland species also declined in that period. This indicates that many, mostly smaller wetlands were still being destroyed or at least turned out not to be suitable any more for some species, e.g. by increasing isolation. With the distribution maps of the 1950s the rapid change in the avifauna of Switzerland is obvious. The use of the same grid as in the two existing breeding bird atlases offers the possibility of comparison for each atlas square.

To our knowledge there is only one other attempt to document historical records for the whole species community on a nationwide level. Holloway (1996) compiled the distribution of breeding birds in Britain and Ireland between 1875 and 1900. However this work did not use the grids of the existing atlases, but for practical reasons county borders instead. Contrary to our project, Holloway (1996) also included the relative abundance. But using original records instead of county avifaunas, this task was not possible in this project. The results of the project will be published in a book in 2011. (Knaus *et al*. in press).

Acknowledgements

This work would not have been possible without the help and the records of many observers, some of them making their observations available especially for this project. I wish also to thank Peter Lüps, Claudia Müller, Bertrand Posse and Niklaus Zbinden for their help with the interviews, Jérôme Guélat, Guido Häfliger and Bernard Volet for preparing the maps, Dominik Hagist, Gesa von Hirschheydt, Rosemarie Küchel, Prisca Mattmann, Bruno Meyer, Daniel Muff, Marianne Parmiggiani, Geneviève Räber, Matthias Schweizer, Ursula Spiess and Elisabeth Taylor for data recording and many other persons for indications of unpublished records and private compilations. Verena Keller, Hans Schmid and Niklaus Zbinden gave valuable comments on earlier drafts of the manuscript.

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MS received: 17.05.2010
Revised MS accepted: 05.11.2010

English name	**Latin name**	**Occupied atlas squares**			**Change**			
		1950–1959	**1972–1976**	**1993–1996**	**1950–1959 vs. 1972–1976**		**1950–1959 vs. 1993–1996**	
					n	**%**	**n**	**%**
Declining species								
Little Bittern	*Ixobrychus minutus*	66	41	40	*-25*	*-46.7*	*-26*	*-49.1*
Common Snipe	*Gallinago gallinago*	28	34	7	*+6*	*+19.4*	*-21*	*-120.0*
Eurasian Curlew	*Numenius arquata*	13	7	2	*-6*	*-60.0*	*-11*	*-146.7*
Little Owl	*Athene noctua*	85	66	8	*-19*	*-25.2*	*-77*	*-165.6*
Eurasian Hoopoe	*Upupa epops*	99	45	12	*-54*	*-75.0*	*-87*	*-156.8*
Woodlark	*Lullula arborea*	50	15	4	*-35*	*-107.7*	*-46*	*-170.4*
Great Reed Warbler	*Acrocephalus arundinaceus*	73	56	48	*-17*	*-26.4*	*-25*	*-41.3*
Icterine Warbler	*Hippolais icterina*	74	74	42	*+0*	*+0.0*	*-32*	*-55.2*
Lesser Grey Shrike	*Lanius minor*	16	2	0	*-14*	*-155.6*	*-16*	*-200.0*
Great Grey Shrike	*Lanius excubitor*	83	33	0	*-50*	*-86.2*	*-83*	*-200.0*
Woodchat Shrike	*Lanius senator*	91	59	8	*-32*	*-42.7*	*-83*	*-167.7*
Ortolan Bunting	*Emberiza hortulana*	22	7	2	*-15*	*-103.4*	*-20*	*-166.7*
Species showing a fragmented distribution already in the 1950s								
Grey Partridge	*Perdix perdix*	51	61	10	*+10*	*+17.9*	*-41*	*-134.4*
Corn Crake	*Crex crex*	50	19	3	*-31*	*-89.9*	*-47*	*-177.4*
Increasing species								
White Stork	*Ciconia ciconia*	3	7	*34*	*+4*	*+80.0*	*+31*	*+167.6*
Black Kite	*Milvus migrans*	110	118	127	*+8*	*+7.0*	*+17*	*+14.3*
Red Kite	*Milvus milvus*	53	60	114	*+7*	*+12.4*	*+61*	*+73.1*
Collared Dove	*Streptopelia decaocto*	42	117	*124*	*+75*	*+94.3*	*+82*	*+98.8*
Stonechat	*Saxicola torquatus*	25	31	33	*+6*	*+21.4*	*+8*	*+27.6*
Fieldfare	*Turdus pilaris*	94	125	126	*+31*	*+28.3*	*+32*	*+29.1*
Northern Raven	*Corvus corax*	31	71	94	*'+40*	*+78.4*	*+63*	*+100.8*

Table 1. Occupied atlas squares in the three periods (selected declining species, species showing a fragmented distribution already in the 1950s and increasing species) and change in the number of occupied atlas squares between 1950–1959/1972–1976 and 1950–1959/ 1993–1996 is given. In contrast to the usual calculations of % change, here the mean of the occupied atlas squares in the 1950s and the 1970s and 1990s, respectively, is used as the denominator, not the number of occupied atlas squares in the 1970s or 1990s. The use of the mean makes increases and declines symmetrical. Maximum decline in the case of extinction is −200%, and maximum increase in the case of colonisation is +200%. The table shows preliminary results.

Bird Census News 2010, 23/1-2: 49-58

Bird species that have significantly changed breeding range on Croatian coastal area: comparison of 30 years old data and recent knowledge

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Abstract. A comprehensive research on the distribution of birds on the Croatian coast took place from 1946 to 1975. We compared data on 106 species from that research with data we collected in recent years and tried to evaluate potential reasons for changes. For 16 species a significant change was recognised. Thirteen species have shown a positive trend and three species a negative one. Potential driving forces for these changes are: reforestation, persecution and hunting pressure, changes in disturbance, agricultural changes, climate change, and changes in behaviour and breeding biology. Our approach could be a starting point for future monitoring.

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Introduction

Recognising changes in distribution and/or abundance of bird species is a starting point in establishing their successful protection. In Croatia, there are tenths of bird species that are known to have changed breeding range and/or abundance in the last decades. However, there is no organised national monitoring of birds scheme running yet. There have been several publications on the distribution of birds in Croatia, all of which encompass smaller geographic scale, scattered across the Croatian territory (Washington 1890, Csörgey 1903, Reiser 1939, Maštrović 1942). Only one comprehensive research was undertaken from 1946 to 1975 on the Croatian coast (Rucner 1998). The data from that research were collected across the whole Croatian coast in habitats influenced by a Mediterranean climate (Figure 1). The area is dominated by the distinctive formation of Dinaric Karst, with a limestone geological base. The primary climatogenic communities have a character of forests, and secondary communities, which have developed from forests by degradation, have a character of scrubland (e.g. maquis), dry grassland, and karstic fields (Horvatić 1971).

In order to get a better idea on what has been going on with the breeding birds in this area we compared Rucner's observations with data we collected in recent years (2004-2009). We have also tried to evaluate the reasons for the observed changes.

Study area

Rucner (1998) carried out his fieldwork in various types of Mediterranean-influenced vegetations characterised by four climazonal associations: *Orno-Quercetum ilicis* Horvatić 1958, *Carpinetum orientalis adriaticum* Horvatić 1939, *Seslerio-Ostryetum* Horvat-Horvatić 1950 and *Seslerio autumnalis-Fagetum illyricum* Horvat 1950. His study area included the whole coastal area up to the most thermophile beech formation on the southern slopes of the coastal mountains (Ćićaria, Velebit, Dinara). The area covers approximately 17 000 km^2.

Figure 1. Map of Croatia with all localities mentioned in the text.
The line represents the border of the research area.

Methods

Field methodology

Rucner did not provide much details on his field methodology and only for a few species he gives quantitative data. For example, he stated the number of singing males of Subalpine Warbler on a 12 km long transect (28.05.1965). From this example and some similar data we can conclude that he used, at least sometimes, long non-standardised transects. Because of the character of the karstic landscape he was not able to design transects in straight lines but had to follow existing footpaths. Observations of nocturnal birds were rare and most of the data on these species result from daytime observations. Rucner probably did few or no night observations. Sometimes he used data from other researchers like Krpan (1970) for Lastovo Island. In addition, Rucner reviewed collections of museums in Zagreb and Metković for some species that were rare visitors to the coastal area. The number of visits per area was never mentioned.

In our research we combined different methodologies: linear transects, non-standardised transects following footpaths, point-counts, territory mapping of nocturnal species and special searches (e.g. inspection of cliffs in search for cliff-breeding species). For colonial breeders we estimated the colony size. Compared to Rucner, we were able to obtain not only presence-absence data but also data on abundance.

Comparison of data

Rucner collected his data from 1946 to 1975 but his observations were not conducted continuously. There are no observations for the Istria region between 1956 and 1974. Such gaps have certainly occurred for other regions too as the whole study area is too large to be continuously covered by one person. He collected absence/presence data for 106 breeding but did not give detailed information on the abundance of a species at a certain locality. Therefore the comparison with the recent data could only be done using the breeding range but not the breeding density. For some species Rucner provided notes on their breeding biology. For example he classified the Blue Rock Thrush as a species of natural cliffs at low elevations only. Giving that the species nowadays also frequently breeds in human settlements and mountain areas, we were could conclude that its breeding density has increased. Another group of species for which we can state changes in abundance are those for which Rucner provides some data (although often general) on their density. For instance, he mentioned the Rock Bunting as a scarce breeder in two small areas. Today the species has become very common and widespread. Due to the lack of good data we could not include nocturnal species in the comparison.

The difference in methodology and research effort (and the fact that we were not able to evaluate this difference) prevented us to perform any statistical tests on the “changes”. In the paper we use the word ‘significant’ not in its statistical meaning, but as a way of expressing more pronounced change.

Results and discussion

For 16 species we detected a clear change (Table 1). For other species the change was less clear, and could be the result of differences in methodology or research effort. Thirteen species have shown positive trends, increasing breeding abundance in the area (two species) or increasing breeding range (eleven species). Four species expanded their breeding range into colder habitats and seven species into warmer habitats. Three species have shown negative trends with decreasing abundance or becoming locally extinct.

Species accounts

Rock Partridge – *Alectoris graeca*

Rock Partridge was once a common breeder all along the coast and on bigger islands. The species is still generally widespread (Radović *et al*. 2005, Radović *et al*. 2006. Budinski 2008, Budinski *et al*. 2008, Budinski *et al*. 2009.) but there are small scale local extinctions possibly caused by over-hunting and reforestation. The same reasons are recognized as the cause of the decline in the remaining of the species range (BirdLife International 2010). Island populations are especially sensitive, although there are occasional invasions from coastal populations. Deliberate local reintroductions from hunting organisations complicate the understanding of the situation. Introduction of the very similar Chukar (*Alectoris chukar*) makes the situation even worse as Chuckar is almost impossible to distinguish from Rock Partridge, unless seen well or heard. On many localities where Rucner considered Rock Partridge as a ''very common'', the species is still present but in small numbers (pers. obs.).

	Species	Trend	Type of change
1	*Alectoris graeca*	-	LE
2	*Athene noctua*	-	LE
3	*Apus pallidus*	++	EC
4	*Merops apiaster*	++	I
5	*Calandrella brachydactyla*	-	LE
6	*Cecropis daurica*	++	EC
7	*Monticola solitarius*	+	EC
8	*Sylvia atricapilla*	+	EW
9	*Cisticola juncidis*	+	EC
10	*Poecile lugubris*	+	EC
11	*Corvus monedula*	+	EC
12	*Passer hispaniolensis*	+++	EC
13	*Fringilla coelebs*	+	EW
14	*Serinus serinus*	+	EW
15	*Coccothraustes coccothraustes*	+	EW
16	*Emberiza cia*	++	I

Table 1. Species that have significantly changed breeding range and/or abundance since the mid-nineties, with trends and type of change*

* EW – species is extending breeding area to warmer habitats
EC – species is extending breeding area to colder habitats
LE – species is declining and/or becoming locally extinct
I – species is increasing in the whole area

Little Owl – *Athene noctua*

Little Owl was common along the coast, more common in the north but also present in the south. Today the species is rare in many areas, and it has disappeared in some areas where Rucner found it (e.g. Mljet Island, Jurinović *et al*. 2007). A decrease of the Little Owl has also been observed in other European countries (BirdLife International 2010). There are two main potential reasons for the decrease in Croatia. One is the loss of open habitats, which is widely recognized (Zmihorski *et al*. 2006, Salek and Schropfer 2008, Thorup *et al*. 2010). The other is the still unproven increase in the population of the Eagle Owl (*Bubo bubo*) and Tawny Owl (*Strix aluco*). For example, the island of Vis is the biggest Dalmatian island without Eagle Owl and is at the same time the only one where the Little Owl is still common. Serrano (2000) found that the Little Owl represents an important part of the Eagle Owl diet. Zuberogoitia *et al*. (2005) and Zabala *et al*. (2006) state that low densities of Little Owl can be the result of predation by Tawny Owl.

Pallid Swift – *Apus pallidus*

Pallid Swift was a scarce breeder on sea cliffs of two outer islands (Mljet and Dugi otok). Those colonies still exist, but we found new colonies along the coast both on sea cliffs and in buildings. In fact, in many coastal areas the species is now more abundant than the Swift (*Apus apus*). The majority of the population occurs in the Dalmatia area where it also breeds in the hinterland. The species is numerous in some coastal towns that Rucner visited (e.g. Makarska, Stari Grad and Korčula) but less common in the hinterland (Sinj, 4-5 pairs among 300-500 pairs of Swift). Although the present population in each of those coastal towns is more than hundred breeding pairs Rucner did not even mention them as breeding sites. The species is less abundant in the Northern Adriatic where the Swift is much more numerous. A possible reason for the expansion could be their adaptation to breed in buildings, something not metioned by Rucner.

Bee-eater – *Merops apiaster*

Bee-eater bred only in several locations in natural habitats; river banks or highly eroded cliffs. The biggest flock Rucner mentioned counted 20 birds. As the species is highly detectable, even during non-breeding season because of its characteristic nesting holes, it is not possible that Rucner failed to find it at more localities. Today it breeds along the coast and on some islands (Susak, Cres, Pag, Hvar, Lastovo, Vis), both in natural habitats and in human settlements, where it uses excavations for house foundations. This type of nest site is a temporal solution. As a result, birds regularly switch to new places from year to year. In high-quality habitat with the lack of breeding sites they also make nest holes on steep grasslands (e.g. Dinara Mt.). Flocks of breeding birds can count more than a hundred and there are colonies with more than 50 pairs on several locations (e.g. Dol on Hvar Island, Turjaci and Vojnić near Sinj, Kijevo near Knin, Ćepić polje in Istria). As a result of rising public awareness and legal protection, the Bee-eater is no more persecuted by bee-keepers as a pest nor by hunters for stuffed bird collections. However, the Bee-eater is considered as a declining species by BirdLife International (2010).

Short-toed Lark – *Calandrella brachydactyla*

Short-toed Lark was a scarce breeder along the coast. Nowadays the species has disappeared from several localities. The most probable explanation for the decrease is the fact that the open short-grass steppe, as its typical habitat, is now mostly overgrown with a scrubland-type vegetation like garigue or with tall-grass vegetations. The species is the first to lose its preferred habitat when grazing of grassland stops and vegetation starts to develope. Relatively stable populations still exist on arid torrential gravel fields, but these habitats are also prone to vegetation succesion. The decline of the species has also been observed in several other countries (Escandell 2004, Vorišek, P. *et al*. 2010). The ecologically similar Calandra Lark (*Melanocorypha calandra*) could be decreasing in the region because of the same reasons. However, this is more difficult to prove because of the small number of localities and the yearly fluctuations.

Red-rumped Swallow – *Cecropis daurica*

Rucner recognised the Red-rumped Swallow as a new expansive breeder in the southernmost part of Croatia. It continued to extend its range along the coast, occupying suitable habitats in the whole of Dalmatia and afterwards in Istria, reaching Slovenia (Mihelič 2000) and the Lika area. Now, the species is widespread, but nowhere common. The expansion has possibly been caused by climate change but also by an increased availability of artificial breeding sites (BirdLife International 2010).

Blue Rock Thrush – *Monticola solitarius*

Blue Rock Thrush was a widespread breeding species along the coast at lower altitudes. It occured in natural habitats and breeding in settlements was recorded as a rare exception. Today, it also breeds even above 1000 m a.s.l., and it regularly breeds in human settlements, even in big cities like Split where there are several pairs in the historical city centre. Recently, it has started to breed in the hinterland where the climate is continental, like in the Lika area, where it prefers to nest on sun-exposed cliffs. Those mountain and hinterland populations are entirely migratory. The reasons for extending the range into colder zones are not clear. The large number of breeding pairs in settlements and houses abandoned during the Croatian War of Independence could have act as a stepping stone in the establishment of urban populations. However, it remains unclear whether this colonisation was caused by changes in breeding ecology (i.e. nesting in houses and buildings) or/and because of the availability of abandoned houses and settlements as a stepping stone.

Blackcap – *Sylvia atricapilla*

Blackcap was a common breeder along the coast in dense deciduous scrublands or forests, and in the dense evergreen Mediterranean vegetation on humid southern islands (Mljet and Korčula). Today it breeds on almost all bigger islands in the dense evergreen habitat, mostly on northern slopes. Reforestation could be the main factor that allowed this species to occupy new areas using newly developed habitats (BirdLife International 2010). In addition, the species is a numerous wintering bird on all islands; so, it is possible that this was another starting point for expansion.

Fan-tailed Warbler – *Cisticola juncidis*

The first proved breeding of the species in Croatia was in Istria in 1974 (Rucner 1998). Although it had not been observed breeding approximately eighteen years before, it was already common at the time of the first breeding record. This population most probably originated from the nearby Northern Italian population. Now, the species breeds all along the coast in small numbers on suitable habitats. It is present even on islands; sometimes on small, isolated patches of suitable habitat that are big enough to support only one pair (e.g. Hvar). The species was also found to increase its range and abundance in other countries (Fernández & Gainzarain 2004, Seoane & Carrascal 2008, BirdLife International 2010).

Sombre Tit – *Poecile lugubris*

Sombre Tit was a localised breeder in the southern coastal area, mostly in forest habitats dominated by Downy Oak (*Quercus pubescens*). Reforestation increased suitable habitats for the species, so it has spread northward into once heavily goat-overgrazed areas. It is possible that the species was always present in those areas, but remained undetected because of its small densities. Now, the species is common in the northernmost Dalmatia and it regularly breeds on southern slopes of the Velebit Mountain. It is very rare in Istria, where it breeds on a small area in the Slovenian part of the peninsula (Geister 1980). It is interesting that Rucner has never found it in Krka river area (although he worked there intensively) where the species is now widespread (Radović *et al*. 2005).

Jackdaw – *Corvus monedula*

Jackdaw was and still is a common breeding species in Istria. In Dalmatia it bred only at one locality. Today, it breeds on several locations all along the Dalmatian coast (Radović *et al*. 2005, pers. obs.) and even in some towns and cities (Split, Dubrovnik and Sinj). The observed range expansion is most likely related to the decrease in human persecution, especially in urban areas. This could have resulted in the colonisation of this areas and facilitated further expansion.

Spanish Sparrow – *Passer hispaniolensis*

Rucner observed Spanish Sparrows on several occasions, but the observations were restricted to the southernmost coastal area. At the time, the colonisation of the species in Croatia had just started, and there was not any certain proof of breeding. In the last 30 years the species has colonised the whole coastal area, and most of the islands and the karstic fields in the Dalmatian hinterland (Radović *et al*. 2005, Budinski 2008, Budinski *et al*. 2008). It also breeds in karstic fields with harsh continental climate in Lika area (pers. obs.). The numerous Croatian population is mostly migratory and uses the central Adriatic Sea flyway. During migration small low-flying flocks are frequently observed on the islands of Mljet, Lastovo and Vis. The reasons for this population change are not clear. A possible reason could be the changes in species ecology that have led to migratory behaviour.

Chaffinch - *Fringilla coelebs*

The Chaffinch was an abundant breeding species in the hinterland but less abundant in the coastal area. It was scarce on the Adriatic islands. Today the Chaffinch is common all along the coast and breeds on all bigger islands. The last island the species colonised was Lastovo (2008-2009, pers. obs.) as this is the furthest big Croatian island. Rucner also considered the species as expanding because he had observed a range extension during his research. Reforestation of the coastal area and islands seems to be the most appropriate explanation for Chaffinch range expansion.

Serin – *Serinus serinus*

The Serin was a scarce breeder, ocurring mostly in hinterland and in some coastal towns. Breeding habitats were edges of pine forests, parks and graveyards with conifer trees. Its habitat is still similar, but it now breeds also in residential areas where suitable habitat is patchily distributed. Nowadays it commonly breeds in the whole coastal area and on some of the islands (Dugi otok, Hvar, Brač and Cres). Several reasons for this increase are changes in agriculture, afforestation with conifers, and decreasing tradition of trapping cage birds. However, in contrary to the trend at the Croatian coast, the species is declining in the rest of Europe (BirdLife International 2010).

Hawfinch – *Coccothraustes coccothraustes*

Hawfinch bred only regularly in the Northern Adriatic while in the southern part it was not recorded until 1972. Now, it is a regular breeder all along the coast in deciduous forests mixed with scrubland and grasslands (Radović *et al*. 2005, Budinski *et al*. 2008). The range extension of the species clearly followed reforestation of the coastal region which is the most obvious reason for this change. According to Rucner and our own data the species seems to prefer the most dense and tallest forests with *Ostrya carpinifolia* and *Carpinus orientalis*, and those habitat types are expanding rapidly. The observed trend in Croatia is consistent with the long-term trend in Europe (Vorišek, P. *et al*. 2010).

Rock Bunting – *Emberiza cia*

Rock Bunting bred only in the vegetation zone of *Ostrya carpinifolia*, mostly in the Northern Adriatic, but also at one locality in Central Dalmatia (Mountain Biokovo). Today it breeds all along the coast from the sea level and even on slopes above the beech forests zone (Radović *et al*. 2005, Radović *et al*. 2006, Budinski 2008, Budinski *et al*. 2008, OIOKON 2008). It also occurs on some islands (Cres, Lastovo) on rocky slopes or rock cliffs surrounded with evergreen Mediterranean vegetation of Aleppo pine (*Pinus halepensis*) and Holm oak (*Quercus ilex*) (pers. obs.). As a species of mosaic landscapes, it also benefits from reforestation of karstic grasslands, but disappears from completely forested areas. The abandonment of rural settlements and traditional, extensive livestock keeping could also have played a role in range size increase, as this results in fewer disturbances, which is an important factor for this ground nesting species. A positive trend is also recorded by Vorišek, P. *et al*. (2010).

Final remarks

The goal of this paper is an provide a first global idea of the trends of various breeding species for which we had enough data available to detect changes in time. This could serve as a kind of starting point for future monitoring. In particular for globally declining species this information is of importance but also for species that are increasing their numbers as they could be potential competitors with endangered species. We could not detect changes for species like the Great Spotted Woodpecker (*Dendrocopos major*), the Black Woodpecker (*Dryocopus martius*) and the Mistle Thrush (*Turdus viscivorus*) although we considered them as species that potentially expanded their range. Nonetheless, giving that these species are low-density breeders and/or inconspicuous, it is possible that they were not recorded before, although they were breeding in the area. In addition, this approach could not detect changes for rare species. For those it is impossible to draw any conclusions about whether they are scarcer now than they were before because there have not been enough records in the past or/and in the recent years. Because of the lack of information on noctural species in Rucner's data, we were not able to include these species into the comparison.

The higher number of species that have increased their range compared to those that decreased could be explained by the fact that it is much easier to detect new breeders than to conclude local extinction. Data about new breeders can be (with certain constraints) obtained from only one field trip but many field trips are necessary to decide that a certain species is not present any more.

This paper shows once more the importance of collecting numerical data in birds' inventories. In our case, a comparison of population and range trends in time was seriously hampered by the lack of quantitative historical data and an good description of the field methodology used by Rucner.

In future researches, priority should be given to the monitoring of the populations of owls on a larger spatial scale, the monitoring of the Short-toed Lark and the Calandra Lark in their known breeding areas, the monitoring of the Rock Partridge (using methodology of counting singing males to avoid confusion with Chukar) and a census of the Pallid Swift and Swift in both natural habitats and in human settlements.

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MS received: 14.05.2010
Revised MS accepted: 01.12.2010

Bird Census News 2010, 23/1-2: 59-69

State of the Breeding Birds in Gediz Delta: Distributions, Abundances, and Changes in Bird populations

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Abstract. The state of the breeding birds of Gediz Delta - West Turkey was identified by BBSs in 2002 and 2006. In 2002, 129 bird species were identified and 93 species were given breeding codes: 47 were confirmed, 23 probable, and 23 possible breeding. In 2006, 142 bird species were identified and 104 species were given breeding codes: 61 confirmed, 24 probable, and 19 possible. Among the identified breeding species, 3 and 4 species were SPEC1, 10 and 11 SPEC2, 34 and 40 SPEC3 in 2002 and 2006, respectively. This study summarises the results of these two Atlas studies.

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Introduction

Breeding Bird Surveys (BBS) or bird atlases have commonly been used for more than three decades and they constantly improve our understanding on species' distributions, population sizes and trends (Gibbons & Donald 2004, 2007). There are numerous BBS studies performed in Europe and some date back to 40 years ago. Conversely to the abundance of studies applied in the European scale and in some other regions like North America, atlas studies in Turkey are, however, very recent and too few in number. The first BBS studies were relatively small-scale, wetland-based studies and were applied by European ornithologists. Among the available studies Brinkmann *et al.* studied (1991) the general situation and the ecological state of the Büyük Menderes Delta which is located on the west coast of Turkey. A year later Hustings & Dijk (1994) studied the birds of Kızılırmak Delta in spring 1992. The standard grid system normally applied in common Breeding Bird Atlas Studies were not applied in these studies. In the first study all the observations were applied in a small number of standard observation points whereas in the latter the study area was sectioned in small study plots. But these during studies equal effort applied complete stratified survey plots were not used and therefore lack of repeatability. Later on Welch & Welch(1998) investigated the breeding birds in the Uluabat Lake and used a similar approach in the methodology. Among these three studies the standard breeding codes (1997) were only used by the last two.

The first study in Turkey which applied a complete grid system and a stratified random technique is the study of Breeding Bird Atlas Survey (BBAS) in Konya Basin (Eken & Magnin 1999). In this study, an area of about 50 000 km^2 was divided into units of 10x10 UTM squares. In each UTM grid, three random points were chosen, 10-minute-long observations were made at each of these points and standard breeding codes were used. Therefore, Eken and Magnin (1999) was formed a kind of baseline methodology for the new BBASs in Turkey. Those new atlas studies can be summarised as Gediz Delta (Izmir) BBAS in 2002 by Onmuş et al. (2009), Kayseri Region BBAS in 2002 by Per *et al.* (2002), Southeast

Anatolia Region BBAS in 2004 by Welch (2004), 19 Mayıs University (Samsun) Campus area BBAS in 2005 by Uker (2006), and Inozu Valley (Beypazarı – Ankara) BBAS in 2007 by Per & Aktaş (2008).

In this study our aim was to identify the state of the breeding birds in Gediz Delta, their distribution, abundance and changes. The first Breeding Bird Atlas Study was applied in 2002 and the results of this study is published by Onmus *et al.* (2009). The Second Atlas study was applied to understand the changes in bird populations four years later in 2006 and this study compares and summarises the results of these two atlas studies.

Materials and Methods

Study Area

Gediz Delta, located on the west coast of Turkey is a wetland of international importance. It satisfies IBA criteria for 28 bird species including important populations of some globally threatened species like Dalmatian Pelican *Pelecanus crispus*, Lesser Kestrel *Falco naumanni*, and Red-Breasted Goose *Branta ruficollis* (Kılıç & Eken 2004). The total area of the delta is about 40 000 Ha consisting of approximately 40% wetland and 60% farmlands and hills. The wetland and its surrounding completing ecosystems comprising a total of 30 500 Ha were chosen for the BBS study. The BBS (atlas) zone is located in two different 50x50 km UTM square grids with Common European Chorological Grid Reference System (CGRS) codes of 35MC3 and 35NCl (European Environment Agency 2008). The atlas boundary coordinates are between 480.000 - 509.000 m E and 4.254.000 - 4.279.000 m N in UTM grid zone 35S (from 38°26'N, 26°46'E to 38°39'N, 27°06'E, WGS84) (Figure 1).

The delta consists of fresh, salty and brackish water ecosystems. Most of the sea-delta boundary is covered with small islets, mud flats, sand flats and sand beaches with *Salicornia* vegetation and seashells. The other key habitats included are coastal marshes, lagoons, Salinas, extensive salt marshes and pastures. In the innermost part of the delta urbanised lands, farmlands and industrial zones are also present. A detailed description of the study area and its habitats is available in the study conducted in 2002 in Gediz Delta by Onmuş *et al.* (2009).

Methodology

The state of the breeding birds was identified by the application of Breeding Bird Surveys (BBS) in May and June in 2002 and 2006. A total of 30 500 ha area comprising of wetland and surrounding completing ecosystems were chosen for the study and was divided into 305 1x1km UTM (Universal Transverse Mercator) grids. To represent each grid, 3 random points separated by at least 300m distance were sampled and surveyed once. Point counts were conducted with a duration of 10 minutes each. The number of point counts applied in a square was reduced in squares having open surface water areas or partial sea, to secure unbiased sampling. Apart from the standard bird atlas recording times, all other casual observations of birds seen and/or heard and their breeding codes were noted separately. The number of individuals for each bird species observed at each points were counted and their breeding evidence were quantified using standard breeding categories and codes (16 codes, subdivided into three categories as Confirmed Breeding, Probable Breeding and Possible Breeding) (Hagemeijer & Blair 1997, Bibby *et al*. 2000). Threats and habitats were recorded for each point. In the 2006 Atlas study, 25 squares having only open water surfaces and/or highly urbanised lands that are not directly or indirectly representing the avifauna of the wetland ecosystem were excluded from the survey.

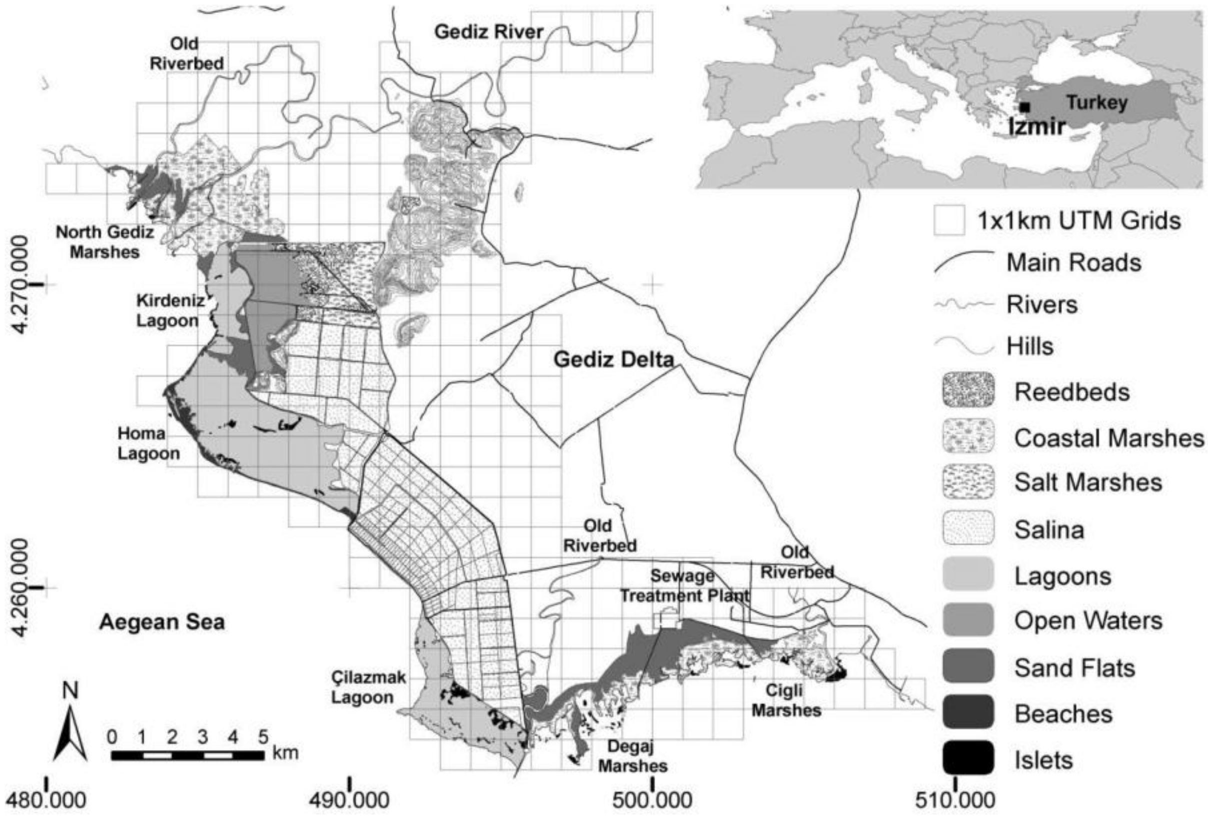

Figure 1. Location of the study area (upper right) within Turkey: A detail of the geography, the main habitats, and the 1x1 km UTM squares.

All the bird atlas forms and casual records were recorded in a MS Access spreadsheet. Distribution maps for each bird species with evidence of breeding were prepared. Breeding evidences and codes were collected separately for the observed/detected bird species in the three census plots in a 1x1 km UTM square, however, the highest breeding code amongst (i.e. highest breeding evidence) given for a bird species in a 1x1 km UTM square was assigned to represent the overall breeding evidence of that species for that square, and was used in the preparation of the distribution maps. Further details of the survey methodology is described by Onmuş *et al*. (2009).

The most common habitat types observed during the two Atlas Studies are presented as a ratio of the total number of UTM squares of which a specific habitat type is observed versus the total number of UTM squares surveyed. The percentage of each habitat observed in 2002 versus 2006 was compared with a Chi-Square Test.

Results

Out of 305 UTM squares, 291 and 266 UTM squares corresponding to 747 and 667 observation points were surveyed in 2002 and 2006, respectively. The numbers, the status of and the number of breeding code species during the two atlas studies are summarised in Table 1.

Year	Total No. of Sp. identified	Total No. of Species (Breeding evidence obtained)	Confirmed Breeding	Probable Breeding	Possible Breeding
2002	129	93 (72.1%)	47 (36.4%)	23 (17.8%)	23 (17.8%)
2006	142	111 (78.2%)	61 (43.0%)	27 (19.0%)	25 (16.2%)

Table 1. Comparisons of number of species identified, number of breeding evidence obtained species and the details of the breeding codes given during the two atlas studies.

The breeding status (Breeding evidence: Confirmed breeding, Probable breeding and Possible breeding), maximum breeding codes (From 1 to 16) and the number of squares with breeding evidence obtained during the two atlas studies are summarised in Table 2.

	2006			2002		
Species name	**Status**	**Max BrC**	**No.Sq**	**Status**	**Max BrC**	**No.Sq**
Tachybaptus ruficollis	C	16	16	C	12	17
Phalacrocorax pygmeus	Ps	1	3	-	-	-
Pelecanus crispus	C	16	4	C	16	4
Ixobrychus minutus	Pr	3	18	Pr	3	9
Nycticorax nycticorax	Pr	3	5	-	-	-
Ardeola ralloides	C	12	4	Ps	1	2
Egretta garzetta	-	-	-	Pr	3	1
Ardea purpurea	Pr	3	1	Pr	3	1
Ciconia ciconia	C	16	9	C	16	9
Platalea leucorodia	Ps	1	1	Ps	1	1
Phoenicopterus roseus	C	16	1	C	13	1
Tadorna ferruginea	C	16	20	C	16	20
Tadorna tadorna	C	16	24	C	15	27
Anas platyrhynchos	Ps	1	2	-	-	-
Anas querquedula	Ps	1	1	Ps	1	1
Circus aeruginosus	Pr	9	9	Pr	9	2
Circus pygargus	Ps	1	1	Ps	1	1
Buteo rufinus	C	13	2	Ps	3	2
Falco naumanni	C	14	7	C	13	14
Falco tinnunculus	C	14	6	C	13	9
Falco peregrinus	-	-	-	Ps	1	2

	2006			2002		
Species name	**Status**	**Max BrC**	**No.Sq**	**Status**	**Max BrC**	**No.Sq**
Alectoris chukar	Ps	1	1	-	-	-
Rallus aquaticus	Ps	2	1	Ps	2	1
Gallinula chloropus	C	16	46	Pr	9	22
Fulica atra	C	16	12	C	16	21
Haematopus ostralegus	C	16	23	C	15	26
Himantopus himantopus	C	15	24	C	16	28
Recurvirostra avosetta	C	13	5	C	13	8
Burhinus oedicnemus	C	13	38	C	16	28
Glareola pratincola	C	13	11	C	13	12
Charadrius dubius	-	-	-	Pr	3	1
Charadrius alexandrinus	C	16	64	C	16	90
Vanellus spinosus	C	16	20	C	12	23
Tringa totanus	C	13	20	Pr	7	15
Larus melanocephalus	C	16	7	C	16	6
Larus genei	Pr	3	2	Ps	1	1
Larus cachinnans	C	16	39	C	16	57
Sterna nilotica	-	-	-	C	13	1
Sterna caspia	C	13	1	C	16	1
Sterna sandvicensis	C	13	1	C	14	4
Sterna hirundo	C	16	36	C	14	49
Sterna albifrons	C	14	26	C	14	46
Chlidonias hybridus	C	14	5	Ps	1	3
Chlidonias niger	Ps	1	1	Pr	3	2
Columba livia	Pr	3	4	Ps	1	2
Streptopelia decaocto	C	14	60	C	13	35
Streptopelia turtur	Pr	7	20	Pr	6	12
Clamator glandarius	Ps	2	1	Ps	1	1
Cuculus canorus	Ps	1	1	Ps	1	1
Tyto alba	Pr	6	1	-	-	-
Bubo bubo	Ps	1	1	-	-	-
Athene noctua	C	13	28	Pr	6	16
Caprimulgus europaeus	Ps	2	2	Ps	2	5
Apus apus	C	13	12	Pr	3	19
Apus pallidus	Ps	1	1	-	-	-
Apus melba	Ps	2	1	-	-	-
Merops apiaster	C	14	60	C	13	35
Coracias garrulus	Pr	3	1	-	-	-
Upupa epops	C	14	2	-	-	-
Dendrocopos syriacus	C	13	14	Pr	3	3
Melanocorypha calandra	C	14	125	C	14	135
Calandrella brachydactyla	C	16	75	C	14	86
Galerida cristata	C	14	228	C	16	225
Alauda arvensis	Pr	3	13	C	14	29
Riparia riparia	Pr	3	8	Pr	6	7

	2006			2002		
Species name	**Status**	**Max BrC**	**No.Sq**	**Status**	**Max BrC**	**No.Sq**
Hirundo rustica	C	16	102	C	16	74
Hirundo daurica	C	13	7	Pr	3	5
Delichon urbica	C	13	23	C	13	21
Anthus campestris	Pr	7	17	C	14	53
Motacilla flava	C	16	100	C	14	78
Motacilla alba	C	14	11	Pr	3	14
Cercotichas galactotes	Pr	9	5	Ps	1	3
Luscinia megarhynchos	Pr	3	10	Ps	2	6
Irania gutturalis	-	-	-	Ps	1	1
Saxicola torquata	Pr	7	4	Pr	3	8
Oenanthe isabellina	C	14	35	C	14	31
Oenanthe oenanthe	C	14	31	C	14	30
Oenanthe hispanica	Pr	3	6	Pr	5	3
Turdus merula	Pr	3	13	Ps	2	7
Cettia cetti	C	14	126	Pr	9	102
Cisticola juncidis	Pr	3	12	Ps	2	3
Acrocephalus schoenobaenus	-	-	-	Ps	2	11
Acrocephalus palustris	Ps	2	7	-	-	-
Acrocephalus scirpaceus	C	14	79	C	14	76
Acrocephalus arundinaceus	C	14	65	C	16	60
Hippolais pallida	C	14	54	C	14	20
Sylvia melanocephala	C	14	10	Ps	2	1
Sylvia curruca	Ps	1	2	-	-	-
Sylvia communis	Pr	5	1	-	-	-
Sylvia atricapilla	Ps	1	1	-	-	-
Phylloscopus collybita	Pr	3	2	-	-	-
Muscicapa striata	Pr	3	4	-	-	-
Panurus biarmicus	C	12	6	C	14	3
Parus lugubris	Ps	2	1	-	-	-
Parus caeruleus	Pr	3	2	-	-	-
Parus major	Pr	3	6	-	-	-
Sitta neumayer	C	13	8	Ps	2	7
Remiz pendulinus	C	16	18	C	14	5
Oriolus oriolus	Ps	2	1	-	-	-
Lanius collurio	Pr	3	15	Pr	3	15
Lanius minor	Pr	3	6	Ps	1	3
Lanius senator	Pr	7	7	Pr	3	4
Garrulus glandarius	Pr	3	5	-	-	-
Pica pica	C	16	112	C	16	98
Corvus corone pallescens	C	14	26	Pr	3	15
Sturnus vulgaris	C	13	3	-	-	-
Corvus monedula	C	14	13	C	13	14
Passer domesticus	C	16	151	C	16	156

	2006			2002		
Species name	**Status**	**Max BrC**	**No.Sq**	**Status**	**Max BrC**	**No.Sq**
Passer hispaniolensis	C	16	52	C	14	44
Passer montanus	C	13	1	-	-	-
Fringilla coelebs	Ps	2	1	Ps	2	1
Carduelis chloris	Ps	2	8	Pr	3	4
Carduelis carduelis	C	16	61	C	13	48
Emberiza cineracea	Ps	2	1	Ps	1	1
Emberiza schoeniclus	Ps	2	1	-	-	-
Emberiza melanocephala	C	15	80	C	14	98
Miliaria calandra	C	14	107	C	14	120

Table 2. All the breeding species registered during the two Atlas Studies (in 2002 and in 2006) in Gediz Delta. (Status C:Confirmed Breeding, Pr:Probable Breeding, Ps:Possible Breeding, Max BrC=maximum breeding code assigned, No.Sq=number of squares that breeding evidence obtained).

Habitat diversity and occurrences at count points

The most common habitat types observed during the two Atlas Studies are shown in Figure 2 as a frequency of observation (percentages among the total number of UTM squares). Habitats with a significant change in their observation frequency (percentage) during the two Atlas Studies are marked with an asterisk in Figure 2.

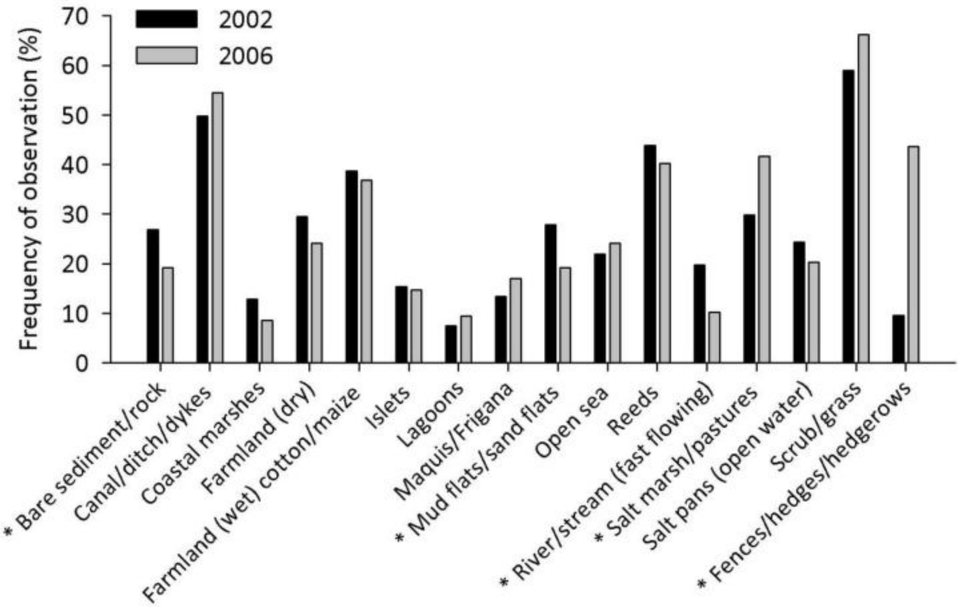

Figure 2. The most common habitat types observed during the two Atlas Studies are given as percentages of the total number of UTM squares a specific habitat observed versus the total number of covered UTM squares. * $p<0.05$

Discussion and Conclusions

During the Atlas studies in 2002 and 2006 a total of 159 bird species were identified and 117 (73.6%) different species were given breeding codes. As a consequence of the methodology applied, some of the nocturnal species might not have been detected commonly during the fieldwork. The details and the comparisons of the breeding codes given during the two atlas studies are summarised in Table 3.

		2006				
	Status	Non-Breeders	Possible	Probable	Confirmed	**Total (2002)**
2002	Non-Breeders	-	12	9	3	24
	Possible	3	9	7	4	23
	Probable	2	2	9	10	23
	Confirmed	1	-	2	44	47
	Total (2006)	6	23	25	61	117

Table 3. Confirmed, probable and possible breeding species and their comparisons in Gediz Delta (2002 vs. 2006)

Among 117 species; 87 (74.4%) species were found in both surveys, while 6 (5.1%) species were only found in 2002, and 24 (20.5%) species only in 2006. Out of the 117 breeding identified species, breeding status did not change for 62 (53.0%) species between the two studies in 2002 and 2006, while it increased for 43 (37.6%) and decreased for 11 (9.4%).

Among the 24 new breeding species in 2006, 3 species were classified as confirmed breeding, 9 as probable, and 12 as possible (Table 1). On the contrary, a total of 6 species that were previously known as breeding in 2002 (Onmuş *et al*. 2009), were not identified as breeding species in 2006.

Among the new identified 24 breeding species, six species; *Phalacrocorax pygmeus, Acrocephalus palustris, Sturnus vulgaris, Oriolus oriolus, Passer montanus*, and *Emberiza schoeniclus* were identified as breeding for the first time both around the region and in the study area. The others were known to breed around the region out of the Atlas Boundary but they were found to breed within the study area for the first time.

The identified non-breeding species in 2006 and that were previously known as breeding species in 2002 were *Egretta garzetta, Falco peregrinus, Charadrius dubius, Sterna nilotica, Irania gutturalis, and Acrocephalus schoenobaenus*. In 2002 *Sterna nilotica* was breeding in a very small population among the breeding colonies of *Sterna hirundo*. In 2006 the lack of breeding activity may either be attributed to missing observation or wrong identification. Regarding *Egretta garzetta* and *Charadrius dubius* we observed that the habitats where these species preferred to used had altered considerably. Similar to *Sterna nilotica*, *Acrocephalus schoenobaenus* observations may be missing.

Many species among the identified breeding birds in Gediz Delta have an "Unfavourable Conservation Status" and therefore qualify as "Species of European

Conservation Concern (SPEC)" (Birdlife International 2004): 3 species were SPEC1 (*Pelecanus crispus*, *Falco naumanni*, *Emberiza cineracea*), 11 SPEC2 (*Ciconia ciconia, Platalea leucorodia, Tringa totanus, Sterna sandvicensis, Caprimulgus europaeus, Coracias garrulus, Oenanthe hispanica, Lanius minor, Lanius nubicus, Emberiza melanocephala, Miliaria calandra*), and 36 SPEC3. According to their threat statuses, 6 species were Vulnerable, 19 species Declining, 4 species Rare, 2 Species Localised, and 20 species were Depleted. Therefore, this inventory confirms that Gediz Delta is an internationally important wetland where both waterbirds and other bird species breed together.

During the atlas studies various threats were identified. Among 291 and 266 squares, 173 (59.5%) and 91 (34.2%) UTM squares were found to be under various threats in 2002 and 2006, respectively. Among the identified threats pollution was found to be the most commonly observed threat and temporary wet grasslands, coastal marshes, salt marsh, and salt pastures were the most threatened habitats. Our results showed that the extension of the threat pressure has relatively decreased in the study area but the threat intensity has increased significantly in some regions. The decrease in the extent of threats may be attributed to the implementation of the Gediz Delta management plan while the increase in the threat intensity in some regions may be attributed to continuing changes in local land use. Since 2002 a Sewage Treatment Plant has been working in the South-eastern part of the Delta (Figure 1) and the produced Sludge (600 tons/day) has been stored around the Plant on coastal marshes, salt marsh and salt pastures. The second reason of pollution is due to the pressure of urbanization. The south-eastern part of the delta is connected to Izmir city and urbanization pressure has been causing both habitat loss and pollution (Onmuş *et al*. 2009, Onmuş & Sıkı 2010). The increase in threat intensity in some regions were observed either to cause a decrease in breeding populations of some of the waterbirds like Sternidae sp. and Charadriidae sp. or to force them to change their breeding grounds (Onmuş & Sıkı 2010).

The increase in the number of breeding species may be explained by several reasons. First; Among the new identified species the reason of breeding of *Sturnus vulgaris, Oriolus oriolus,* and *Passer montanus are* thought to be due to the effects of global warming (the data is being prepared for publication). Second; Despite the increase in threat intensity, the decrease in the extent of threats might have also resulted in an increase in the number of breeding species. This observation is also well correlated with an increase in the observation frequency of the habitat Fences/hedges/hedgerows (Figure 2). In 2004, about more than 10 km long roadside of the one of the main road located in the south-eastern part of the delta and forming the border between natural habitats and settlements were cleaned from illegal dumping and bordered with fences by the Izmir Bird Paradise conservation and development Union. This activity was resulted in a decrease in the threat pressure on these habitats and a increase in habitat quality. Other reason of the increase in some of the breeding cryptic species might be attributed to the more developed skills of identification or the researchers' increasing acquaintance with the study area in 2006, and these might have led to bias in identification of some of the new species.

In this study relative abundance values (frequency of occurrences in UTM squares) were calculated but not presented along with the data here. Moreover the number of breeding pairs in all the colonial breeding waterbirds were censused and some of the data is presented elsewhere (Onmuş & Sıkı 2010). In the future we plan to analyse and present more reliable density values based on the collected data on fixed radius point distance bands in point counts.

Acknowledgements

We would like to thank the members of Ege Bird Watching Club and some other birdwatchers who have voluntarily contributed in the implementation of the Breeding Bird Atlas Surveys. We also would like to acknowledge Doğa Derneği (BirdLife - Turkey) and Izmir Bird Paradise Conservation and Development Union who partially financed the 2006 Atlas project. Finally we would like show our appreciation to the Provincial Directorate of the Nature Conservation and Natural Parks of Izmir of the Turkish Ministry of Environment and Forestry, Izmir Bird Paradise Union for providing permission and support for the Breeding Bird Atlas Projects.

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MS received: 25.05.2010
Revised MS accepted: 25.10.2010

Bird Census News 2010, 23/1-2: 71-77

Abundance and community composition of waterbirds during a yearly cycle in coastal wetlands of Friuli Venezia Giulia and Emilia-Romagna (NE Italy), Southern Croatia and Albania

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Abstract. We carried out coordinated waterbird monitoring scheme within the frame of the Interreg ANSER project during two years in several coastal wetlands of NE Italy, S Croatia and Albania. Waterbirds and some wetland-related raptor species were censused twice per month through Daylight Counts. The community structure of each study area was analysed for the first yearly cycle. The communities in the three areas were described by several parameters. The monthly numerical fluctuations and the species composition of the taxonomic groups most represented are described.

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Introduction

A coordinated monitoring programme was developed within the transnational cooperation project ANSER “Ecological role of the coastal wetlands in northern Adriatic, for the stop over and the wintering of water birds: guidelines for the preservation and the management of the coastal natural marine heritage”. The main aim was to build up a shared instrument for monitoring and managing waterbird populations around the Adriatic Sea, mainly in North-East Italy, Southern Croatia and Albania.

Material and Methods

Waterbird surveys were carried out according to standard methodologies (Serra *et al.* 1997, Gibbons & Evans 1998, Madsen 1998, Bibby *et al.* 2000 and Brown *et al*. 2005) between June 2006 (December 2006 in Croatia) and May 2008. For location of the study areas and other details see Fig. 1 and Table 1.

Birds were censused by “Daylight Ground Counts” twice per month (at spring high tides in tidal areas); these were integrated with Aerial Counts in Friuli Venezia Giulia only. Large wetland complexes were surveyed in one day to decrease sampling biases, while smaller and isolated sites were usually surveyed the following day.

Figure 1. General localization of the study area of ANSER project, including NE Italy (Friuli Venezia Giulia and Emilia-Romagna), S Croatia (Split-Dalmatia and Dubrovnik-Neretva Counties) and Albania. Project area in Friuli Venezia Giulia (oval frame) includes the whole coastal wetland system, which is composed by 39 wetlands.

The community structure of each study area was analysed for the period June 2006 - May 2007 (December 2006 - November 2007 in Croatia), taking the highest value from the two monthly counts available for each species. For richness, the overall number of species observed in a month was used. The bird community of each area was described by the following parameters: monthly richness (r), average monthly richness (r’), annual total richness (R), abundance (n), relative abundance (pi), number of dominant species (Nd), number of subdominant species (Nsd, *sensu* Turcek 1956), dominance index (DI; Wiens 1975), diversity (H’; Shannon & Weaver 1963), evenness (J’; Pielou 1966) and evenness index ($e^{H'}$; Pielou 1969), used as an Index of Utilization of the wetland over Time (IUT, *sensu* Casini *et al.* 1992).
The monthly numerical fluctuations and the species composition of main waterbird communities were described.

Results and discussion

Studied wetlands were characterized by the presence of different habitats and very variable sizes, ranging from 50 to 34 434 ha.

Summarized results about community composition and abundance are shown in Tab. 1, including the list of dominant and subdominant species. Monthly overall abundance values of most represented waterbird taxonomic groups for each study area are reported in Figs. 2-9.

The highest values of waterbird monthly abundance were found in Italy, in decreasing order, in Friuli Venezia Giulia wetlands, Saline di Cervia and Ortazzo-Ortazzino; high values were recorded also in Albanian wetlands (Karavasta and Patoku), while abundance was low in Croatian sites, in Prološko blato in particular.

The highest values of annual richness were found in the widest area (Friuli Venezia Giulia), followed by those of Emilia-Romagna wetlands and, quite surprisingly considering their relatively small extension, by Albanian wetlands. The lowest values were recorded in Pantan and, in particular, in Prološko blato, both Croatian wetlands. The wide estuarine Neretva area showed also quite low richness values.

Waterbird diversity reached the highest values in Albanian sites, but it is rather high in Italian wetlands too, especially in Ortazzo-Ortazzino, while the lowest values were recorded in Prološko blato (Croatia).

A total amount of 25 dominant and/or subdominant species was found, including: *Anas penelope* (dominant in 4 sites; subdominant in 3 sites), *Chroicocephalus ridibundus* (4; 3), *Fulica atra* (6; 0), *Anas crecca* (5; 1), *Anas platyrhynchos* (4; 1), *Calidris alpina* (2; 3).

The 5 species with highest IUT for each study area are shown in Figs. 10-17. IUT values lower than 6-7, indicated a strong periodicity due, for example, to a wide range variation of water levels (as in Prološko blato and Pantan). Species with IUT>6 have been recorded in several sites: *Ardea cinerea* (6 sites); *Egretta garzetta*, *Chroicocephalus ridibundus*, *Phalacrocorax carbo*, *Numenius arquata* (5 sites); *Tachybaptus ruficollis* and *Tringa totanus* (4 sites).

Conclusion

Despite the great variation in size and habitats, surveyed wetlands were featured by the presence of some species which can be assumed as representative of the Adriatic waterbird community. They were both migrant (*Fulica atra, Anas crecca, Anas Penelope, Calidris alpina*) and partially resident species (*Chroicocephalus ridibundus, Anas platyrhynchos*).

The highest values of richness and biodiversity were observed in the widest and/or more diversified wetlands. Albanian wetlands had the highest biodiversity values and very high richness values despite their relatively small sizes, to indicate that they are probably located close to the divide between two biogeographic regions, and hence hosting two different waterbird communities. These results support the need of a shared management of Adriatic waterbird populations.

Acknowledgements

Thanks to all the teams which collaborated in collecting and processing data.

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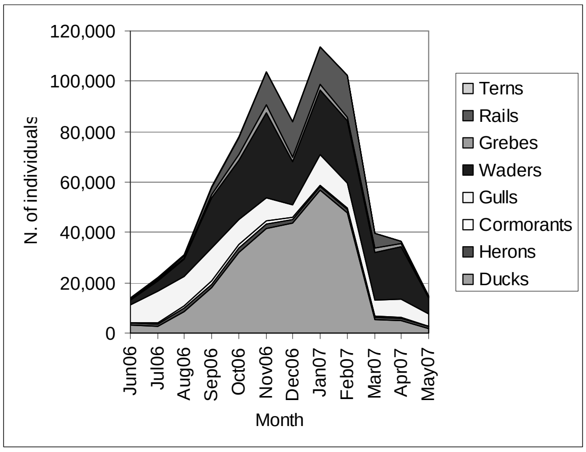
MS received: 13.05.2010
Revised MS accepted: 04.10.2010

Figure 2a-h. Monthly overall abundance values of most represented waterbird taxonomic groups during a yearly cycle in 8 different sites (from left to right and up to down) with most abundant species. **2a**. Friuli Venezia Giulia: *Calidris alpina* (total annual abundance: 146,442), *Anas penelope* (134,469), *Fulica atra* (75,821), *Anas platyrhynchos* (72,543), *Larus michahellis* (68,515). **2b**. Saline di Cervia (Emilia-Romagna): *Larus michahellis* (total annual abundance: 48,100), *Calidris alpina* (27,334), *Larus ridibundus* (13,318), *Phoenicopterus roseus* (6,473), *Recurvirostra avosetta* (7,575). **2c**. Ortazzo e Ortazzino (Emilia-Romagna): *Fulica atra* (15,702), *Anas platyrhynchos* (13,418), *Anas crecca* (5,301), *Larus ridibundus* (4,301), *Anas penelope* (2,846). **2d.** Prološko blato (Croatia): *Podiceps cristatus* (180), *Fulica atra* (143), *Anas clypeata* (103), *Anas platyrhynchos* (73), *Tachybaptus ruficollis* (66). **2e**. Neretva River Mouth (Croatia): *Anas crecca* (1,025), *Phalacrocorax aristotelis* (810), *Anas penelope* (363), *Egretta garzetta* (195), *Larus ridibundus* (133). **2f**. Pantan (Croatia): *Fulica atra* (843), *Anas crecca* (351), *Larus ridibundus* (85), *Anas platyrhynchos* (66), *Larus michahellis* (63). **2g**. Karavasta (Albania): *Fulica atra* (30,560), *Anas penelope* (9,446), *Larus ridibundus* (8,006), *Anas crecca* (5,868), *Anas clypeata* (4,850). * Census not performed for logistic impediment. **2h**. Patoku (Albania): *Larus ridibundus* (8,155), *Fulica atra* (3,018), *Anas penelope* (2,332), *Vanellus vanellus* (2,192), *Numenius arquata* (1,424).° Census not performed for habitat unsuitability (dried-up wetland).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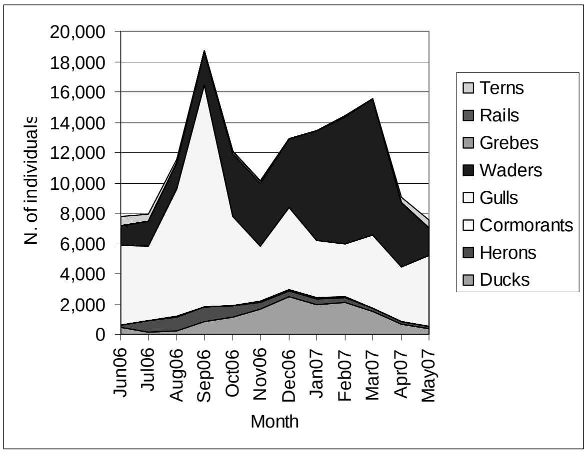
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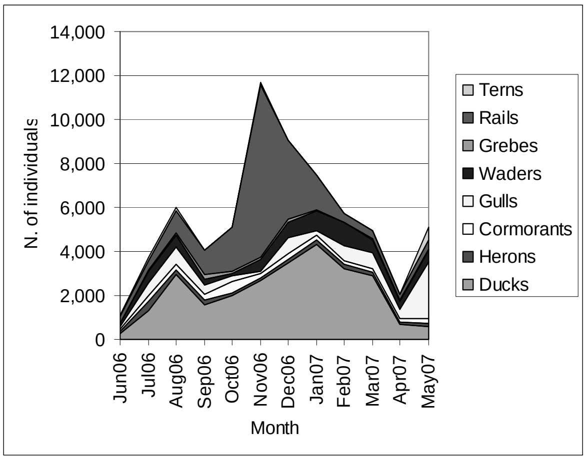
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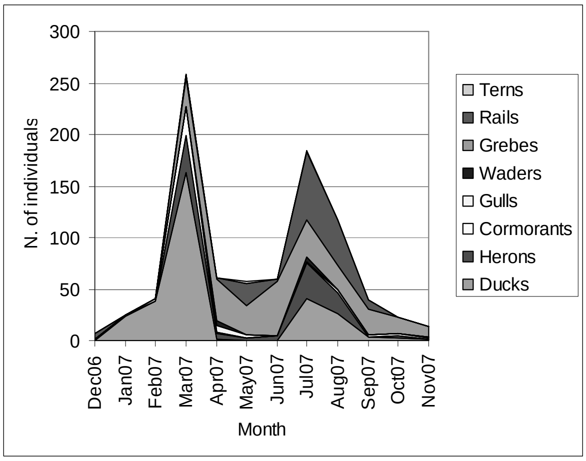
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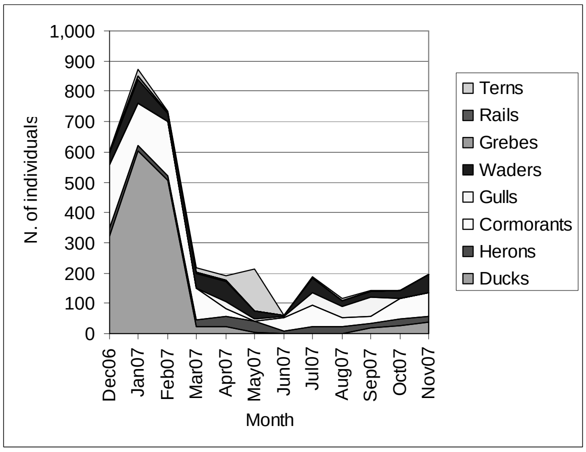
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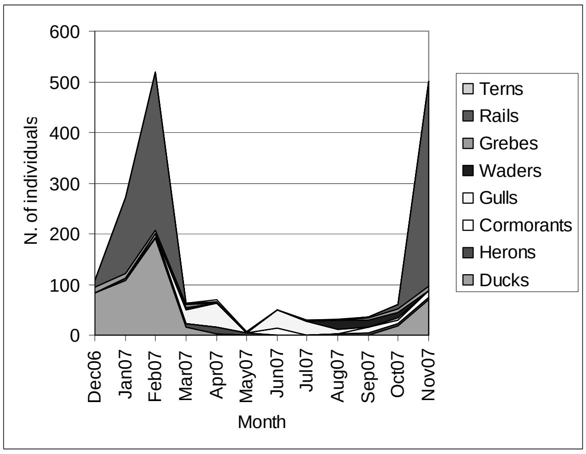
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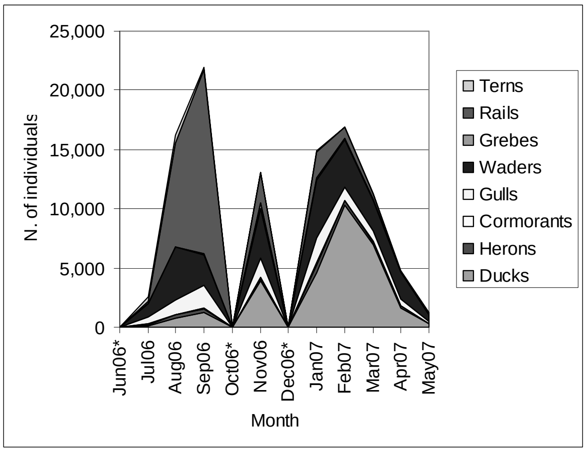
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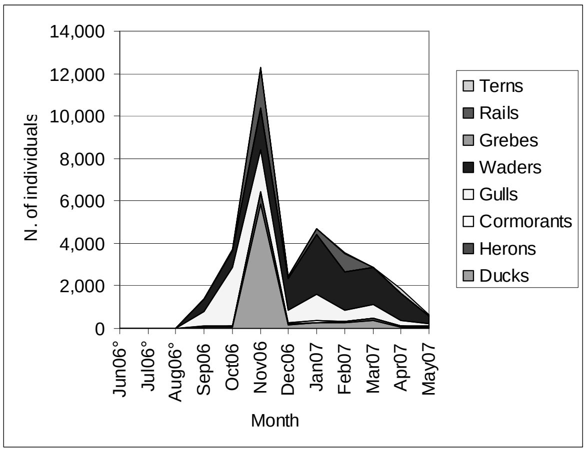
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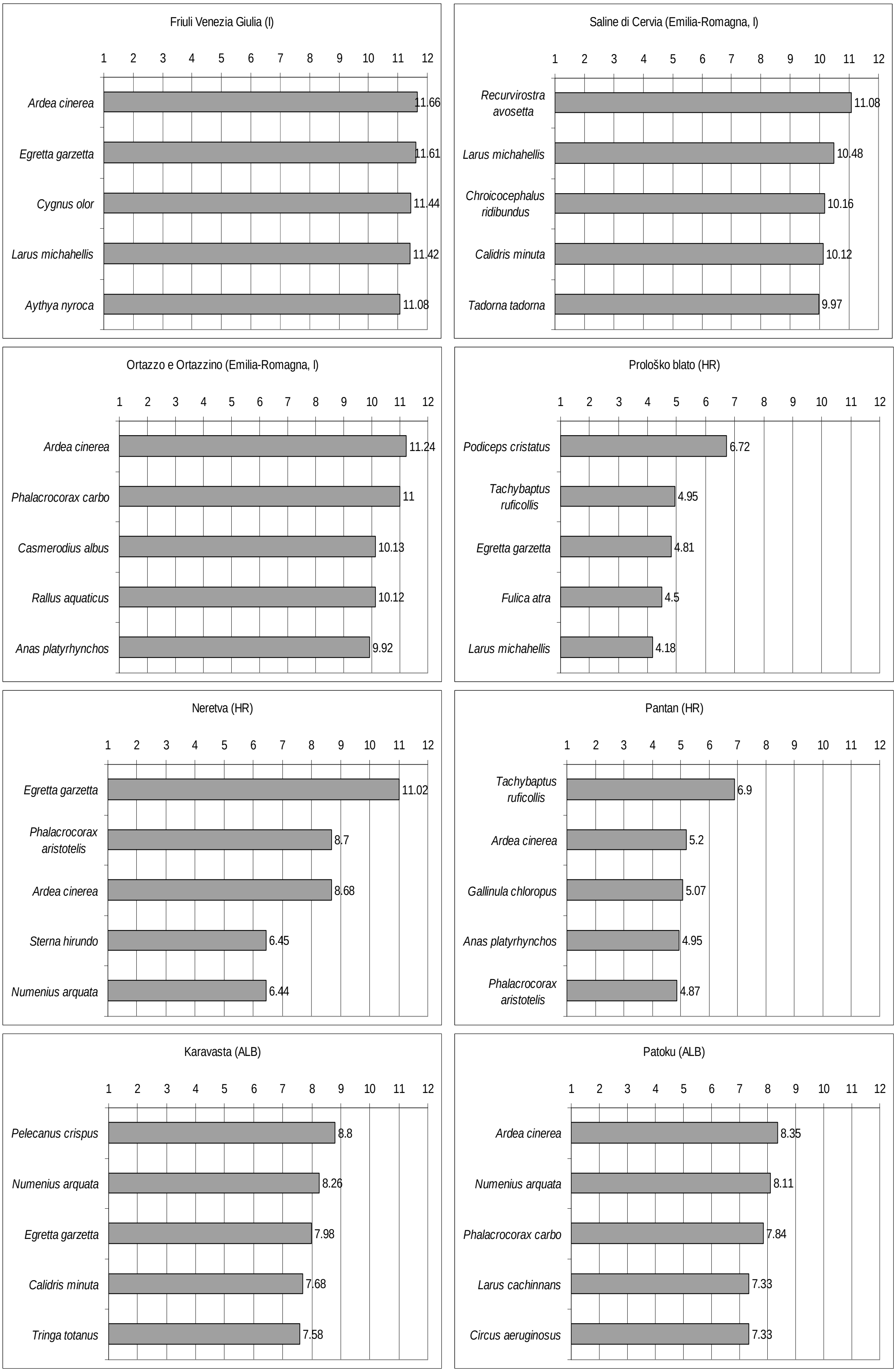

Figure 3: Waterbird species with highest Index of Utilization (IUT) in each study area (see Figures for names 2).

Wetland/Region (Country)	**Friuli Venezia Giulia (I)**	**Saline di Cervia, Emilia-Romagna (I)**	**Ortazzo e Ortazzino, Emilia-Romagna (I)**	**Prološko blato (HR)**	**Neretva (HR)**	**Pantan (HR)**	**Karavasta (ALB)**	**Patoku (ALB)**
Wetland area (ha)	34,434	828	1,418	1,035	24,915	50	5,450	2,194
Proportional area (%)	49.0%	1.2%	2.0%	1.5%	35.4%	0.1%	7.7%	3.1%
Mean* monthly richness (r)	52÷71	27÷42	30÷53	2÷12	7÷22	2÷12	41÷64	13÷43
Total annual richness (R)	98	65	77	23	45	27	85	76
Total annual abundance	697,351	144,910	65,956	885	3,721	1,747	97,100	28,016
Mean* monthly abundance (n)	13,971÷113,896	7,636÷19,532	1,111÷11,696	7÷184	61÷872	6÷501	1,181÷21,177	518÷9,645
Dominant species (sorted by systematic order)	*Ana pen, Ana pla, Ful atr, Cal alp*	*Cal alp, Chr rid, Lar mic*	*Ana cre, Ana pla, Ful atr, Chr rid*	*Ana cre, Ana pla, Ana cly, Ard cin, Tac ruf, Ful atr*	*Ana pen, Ana cre, Pha pyg*	*Ana cre, Ful atr*	*Ana pen, Ana cre, Ana pla, Ful atr, Chr rid*	*Ana pen, Ful atr, Van van, Chr rid*
Subdominant species (sorted by systematic order)	*Ana cre, Plu squ, Num arq, Chr rid, Lar mic*	*Pho ros, Rec avo*	*Tad tad, Ana pen, Pha car, Lar mic*	*Ana pen, Pha car, Egr gar*	*Egr gar, Pha ari, Cal alp, Num arq, Chr rid, Chl nig, Ste hir*	*Ana pen, Ana pla, Pha pyg, Tac ruf, Gal chl, Chr rid, Lar mic*	*Ana cly, Van van, Cal min, Cal alp, Tri tot*	*Num arq, Cal min, Cal alp, Tri tot*
Diversity (H')	2.13÷2.64	1.79÷2.42	1.83÷2.88	0.17÷1.92	1.13÷2.71	0.58÷2.27	1.33÷3.33	0.65÷3.13

Table. 1. Summary table of main parameters describing waterbird community in the study area, including list of dominant and subdominant species for each study area (Ana cly: *Anas clypeata*; Ana cre: *Anas crecca*; Ana pen: *Anas penelope*; Ana pla: *Anas platyrhynchos*; Ard cin: *Ardea cinerea*; Cal alp: *Calidris alpina*; Cal min: *Calidris minuta*; Chl nig: *Chlidonias niger*; Chr rid: *Larus ridibundus*; Egr gar: *Egretta garzetta*; Ful atr: *Fulica atra*; Gal chl: *Gallinula chloropus*; Lar mic: *Larus michahellis*; Num arq: *Numenius arquata*; Pha ari: *Phalacrocorax aristotelis*; Pha car: *Phalacrocorax carbo*; Pha pyg: *Phalacrocorax pygmeus*; Pho ros: *Phoenicopterus roseus*; Plu squ: *Pluvialis squatarola*; Rec avo: *Recurvirostra avosetta*; Ste hir: *Sterna hirundo*; Tac ruf: *Tachybaptus ruficollis*; Tad tad: *Tadorna tadorna*; Tri tot: *Tringa totanus*; Van van: *Vanellus vanellus*). * Range (highest and lowest values).

Bird Census News 2010, 23/1-2: 79-85

How to assess changes in bird distribution between successive atlas projects with different grids and survey coverage

Viesturs Ķerus, Ainārs Auniņš & Māris Strazds

Abstract. Three national level breeding bird atlases carried out in Latvia are compared to explore the possibilities of evaluating change in bird distribution if grid and survey coverage differ among studies. It is advised to equalise the grids as much as possible and analyse only the grid squares with similar survey coverage in all projects. However, each species should still be treated separately as overall differences in survey coverage is not the only factor masking or exaggerating the change in bird distribution.

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Introduction

Since the first breeding bird atlas (Sharrock 1976) the general methodology of a "bird atlas" has become very wide-spread and many atlases on different scales – local (e.g. Strazds *et al.* 1998, Strazds *et al.* 1999, Strazds & Ķuze 2006), regional (e.g. Walasz 1992, Jusys *et al.* 1999), national (e.g. Priednieks *et al.* 1989, Schmid *et al.* 1998, Hustings & Vergeer 2002) and continental (Hagemeijer & Blair 1997) – have been published. Although the methods have become more advanced (Gibbons *et al*. 2007), the general goal of such studies has remained the same – documenting the distribution of birds. With atlas studies carried out in the given territory repeatedly, temporal dimension is added and it is possible to evaluate also the changes in the distribution. The general rule to study changes in any parameter over the time is to carry out the research with the same methods, but given the large time-scale on which atlas studies are repeated, this is not always possible. Some options in dealing with such problems are explored in this paper.

Breeding bird atlases in Latvia

For the purposes of this paper we compare three breeding bird atlases carried out on the national scale in Latvia: the first Latvian Breeding Bird Atlas (1980-1984; Priednieks *et al.* 1989) European Breeding Bird Atlas (1985-1989; data for Hagemeijer & Blair 1997) and the second Latvian Breeding Bird Atlas (2000-2004; LOB unpublished). The names of the three studies are hereafter abbreviated: LBBA 1980-1984, EBBA 1985-1989 and LBBA 2000-2004. Details regarding the differences between these studies are shown in Table 1.

It can be seen that the first two atlases are very different from the last one thus posing problems in comparing the data directly. Firstly, the grid not only differs in the square size but also in the projection used, which means that squares do not overlap precisely. Secondly, the finer resolution and larger number of participants leads to much better overall survey coverage in the third study. We tried several approaches dealing with these problems. First of all, all distribution data with recorded codes of breeding probability (only possible, probable or confirmed breeding; Hagemeijer & Blair 1997) were transformed to presence/absence, this way minimising the impact of interpretation by the observer on the data.

Abbreviation	Atlas study	Time period	Grid size	Grid projection	Number of squares	Number of observers
LBBA 1980-1984	First Latvian Breeding Bird Atlas	1980-1984	10x10 km	UTM	739	130
EBBA 1985-1989	European Breeding Bird Atlas	1985-1989	10x10 km*	UTM	739	136
LBBA 2000-2004	Second Latvian Breeding Bird Atlas	2000-2004	5x5 km	LKS-92 TM	2785	325**

Table 1. Differences between three nation-wide atlas studies carried out in Latvia.
* Although data in Hagemijer & Blair (1997) were used in 50x50 km squares, data were collected in Latvia at a finer scale. ** Observers submitting only anecdotal records are not included

Transferring the data in LKS-92 TM grid to UTM grid

For the rarest species additional information was gathered including precise coordinates of the observation. This allows transferring the point data to any grid of interest, in this case – the UTM grid of 10x10 km squares. It must be noted, however, that in this way we lose the information on differences in survey coverage, thus probably having an incorrect picture of changes in distribution. This is true in the example on Hoopoe *Upupa epops* (Figure 1). Although the distribution seems to have increased, there is no reason to believe it

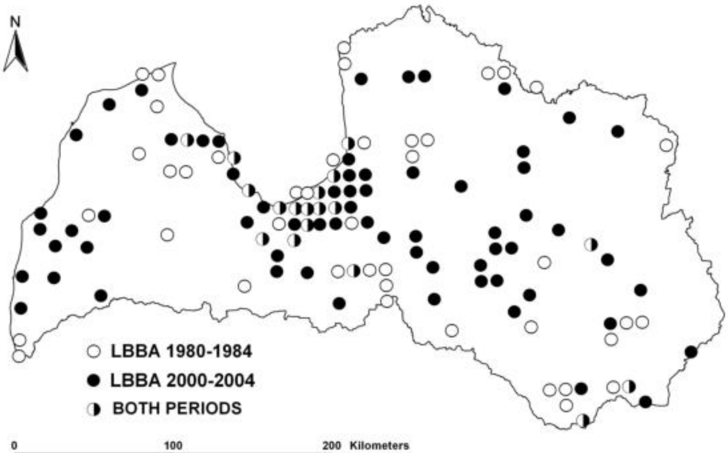

Figure 1. Changes in distribution of Hoopoe *Upupa epops* in Latvia between two national atlas projects

is true as special attention was paid to this species due to a campaign by Latvian Ornithological Society (LOB), which accounts for about half of recordings for this species in 2000-2004 (Ķerus 2005). In any case, only a minority of species (i.e., the ones with data on exact locations) can be analysed this way. For other species separate maps for each study period can be compared, equalising the grids as described below.

Equalising the grids

If 5x5 km squares used in the LBBA 2000-2004 are merged into 10x10 squares, the total number of squares – 740 – is almost identical to the first two studies (Table 1). This has two main advantages: 1) visual comparison of distribution maps is more comprehensible; 2) number of squares can be used as an indicator of changes in distribution. An example is given in Figure 2A. It must be noted, however, that due to the great differences in survey coverage the general pattern with the number of squares with the species recorded being the lowest in the EBBA 1985-1989 and the largest in the LBBA 2000-2004 is the same for most species.

Species	LBBA 1980-1984	EBBA 1985-1989	LBBA 2000-2004
White Stork *Ciconia ciconia*		x	x
Common Buzzard *Bute buteo*	x	x	x
Wood Pigeon *Columba palumbus*			x
Cuckoo *Cuculus canorus*	x		x
Skylark *Alauda arvensis*	x	x	x
Barn Swallow *Hirundo rustica*	x	x	x
House Martin *Delichon urbica*	x		
Tree Pipit *Anthus trivialis*	x	x	x
White Wagtail *Motacilla alba*	x	x	x
Robin *Erithacus rubecula*	x	x	x
Whinchat *Saxicola rubetra*	x	x	x
Blackbird *Turdus merula*	x	x	x
Song Thrush *Turdus philomelos*	x	x	x
Whitethroat *Sylvia communis*		x	x
Garden Warbler *Sylvia borin*		x	
Wood Warbler *Phylloscopus sibilatrix*	x	x	x
Chiffchaff *Phylloscopus collybita*	x	x	x
Willow Warbler *Phylloscopus trochilus*	x	x	x
Great Tit *Parus major*	x	x	x
Magpie *Pica pica*	x		
Hooded Crow *Corvus cornix*	x	x	x
Raven *Corvus corax*	x	x	x
Starling *Sturnus vulgaris*	x	x	x
Chaffinch *Fringilla coelebs*	x	x	x
Yellowhammer *Emberiza citrinella*	x	x	x
TOTAL	21	21	22

Table 2. The most common species recording of at least 75% of which was used as an indicator of sufficient survey coverage of an atlas square.

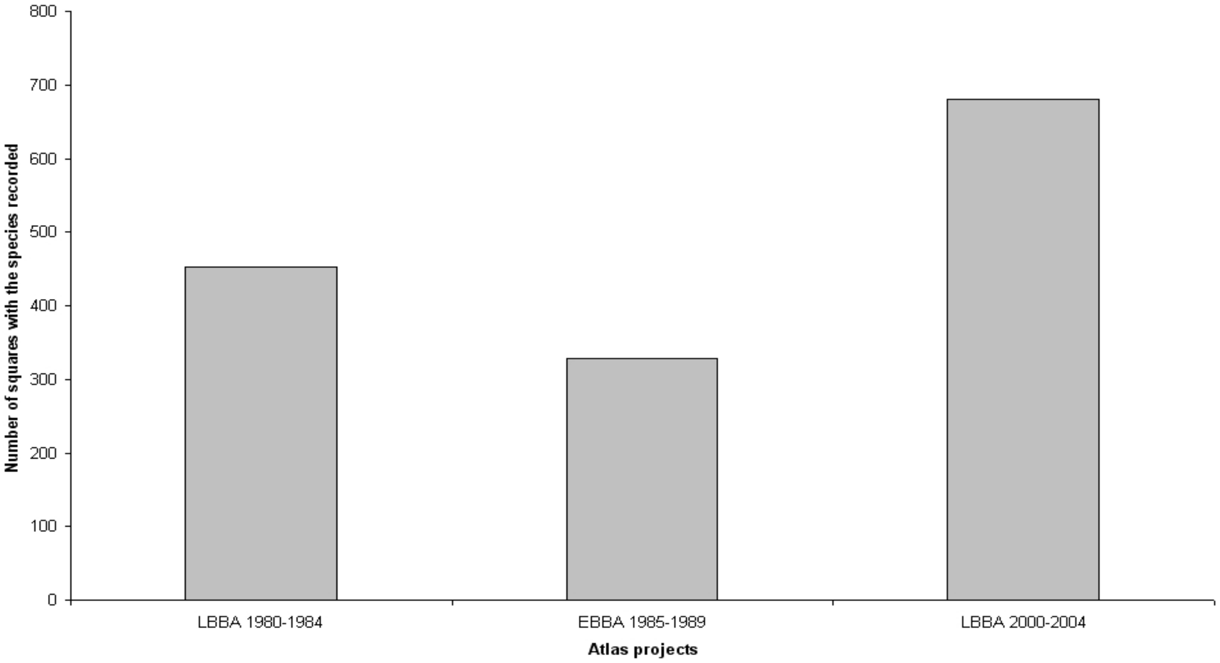

Figure 2A. Changes in the absolute number of 10x10 km atlas squares with Corncrake Crex crex recorded in three atlas studies in Latvia

To evaluate the survey effort and changes in this parameter, one method has to be used in all cases, therefore, we were limited by the lowest quality data we had in our studies. For the LBBA 1980-1984 we have only the list of species (with their breeding probabilities, which were not used, as explained before) for each square. The approach to evaluate the survey coverage used by Priednieks, Strazds *et al.* (1989) was to select the most common species (10% of the total number of species recorded) and consider squares with at least 75% of these species (i.e., at least 16 of the most common species in LBBA 1980-1984 and EBBA 1985-1989 and at least 17 of the species in LBBA 2000-2004) recorded covered sufficiently. We used this method to evaluate the survey coverage also in the other two studies (Table 2). The number of species used depends on the total number of species recorded in each study as top 10% commonest species were used. It must be noted that due to the fact that most of the most common species are forest or farmland birds, this approach gives no indication of survey coverage of specific habitats (bogs, wetlands etc.).

We further analysed only the sufficiently covered squares. There are 618 of them in the LBBA 1980-1984, 411 in EBBA 1985-1989 and 696 in the LBBA 2000-2004. Therefore, we have different number of squares in each study and proportions of squares with the species recorded have to be used to evaluate changes in distribution, which allows seeing a clearer picture of change (Figure 2B).

Statistical comparison

Our main interest was in measuring the significance of the changes. For this task we found that 2x2 table approach (Table 3) is most suitable for our data. Although it does not allow spatially explicit analysis of changes, it allows avoiding problems caused by incompatible reference grids.

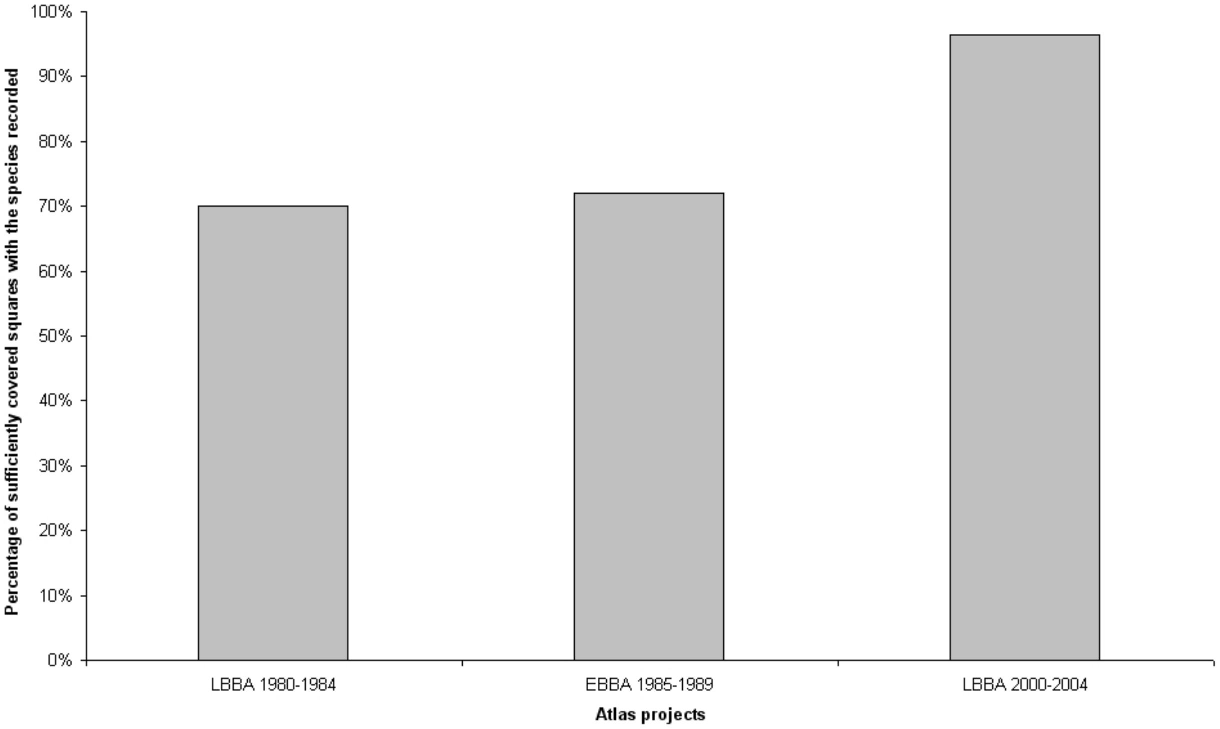

Figure 2B. The proportion of sufficiently covered squares with Corncrake *Crex crex* recorded in three atlas studies in Latvia

Sokal & Rolf (1995) prefer G-test of independence for analysis of 2x2 tables. However, this test becomes problematic when we have a species with the number of squares being 0 in any of the cells, as the natural logarithm needed in this test cannot be calculated for 0. This is why we chose chi-square test of independence and it gave almost identical results to those obtained by G-test for those tested species which allowed use of both tests.

In order to give the complete picture it would be advised to use all approaches combined: distribution maps let see the spatial change, analysis of 2x2 tables shows the significance of the changes and comparison of the proportions of squares with species recorded clearly communicates the direction of change.

	Number of squares	
	Not recorded	Recorded
LBBA 1980-1984	185	433
LBBA 2000-2004	25	671

χ^2=169.18.10, p<0.01

Table 3. Changes in distribution of Corncrake *Crex crex* between LBBA 1980-1984 and LBBA 2000-2004 analysed by chi-square test of independence.

Problem situations

Although the approach explained above could be appropriate for general evaluation of changes in breeding distribution for most of the species, some problems are posed by the fact that in this way only differences in overall survey coverage are controlled for.

Most serious problems could be:

1. Survey coverage of specific habitats or sites differing from the general survey coverage.
2. Improved knowledge on species.
3. Species with restricted distribution.

Example for the first case in the case of Latvia is survey coverage of bogs. During 1980-1984 special attention was paid to cover all largest bogs (Priednieks *et al.* 1989), but in 2000-2004 although the overall survey coverage has improved, less attention has been paid to targeted survey of bogs, thus leading to under-representation of bog specialist species, e.g., Golden Plover *Pluvialis apricaria*.

The most striking example of species knowledge on which has been improved is Pygmy Owl *Glaucidium passerinum*. It was thought to be a very rare species, but correct approach to recording this species (using playback during afternoon) lead to revelation that this is in fact one of the most common owl species in Latvia. The case of Pygmy Owl is simple because the improvement in knowledge is well-recorded, but it can be presumed that there are also other species which are on average better recorded with improving knowledge.

The last case – species with restricted distribution – is as theoretical one, but it must be noted that if the distribution of a species is restricted to squares sufficiently covered in all atlas periods, changes in overall survey coverage will produce artificial changes in distribution even if the distribution has not changed.

Conclusions

To compare changes in bird distribution equalization of squares helps even if it is not complete. Only squares with similar coverage concerning the species or group of interest should be compared. Analysing 2x2 tables, chi-square test of independence is more suitable for evaluation of changes in distribution than G-test of independence as it allows working with tables containing zeros as cell values. Differences in overall survey coverage is only one factor exaggerating (or masking) changes in bird distribution, therefore, each species has to be assessed individually.

Acknowledgements

The study was funded by the University of Latvia and ESF in the project “Support for Doctoral Studies in University of Latvia”. Data of breeding bird atlases were supplied by Latvian Ornithological Society (LOB). An anonymous referee is acknowledged for suggesting improvements to the paper.

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MS received: 17.06.2010
Revised MS accepted: 29.10.2010

Bird Census News 2010, 23/1-2: 87-97

Developing a national Farmland Bird Index for Austria

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Abstract. This paper documents the species selection for the Austrian Farmland Bird Index used for the EU's Rural Development Programme and shows the resulting indicator compared to indicators based on two other species selections: the species list for this indicator proposed by the European Union, and the regional farmland bird list for 'continental' Europe developed by the European Bird Census Council. We summarize arguments in favour and against selection of an initial long-list of potential indicator species. The decisions were based on the criteria of strong linkage to farmland habitat and farming practices, reliable recording, frequency of occurrence in counts, reliable breeding status, and lack of persecution or exploitation by humans, using mainly data of breeding bird population estimates for Austria and of the Austrian Common Bird Monitoring Scheme. The final Austrian indicator species selection differs from both supranational species lists. The trends of all three multi-species indicators are correlated significantly to each other, but have different slopes, and hence suggest different rates of decline. We discuss aspects of species selection and resulting differences in trends.

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Introduction

The Farmland Bird Index is used by the European Union's member states to evaluate measures implemented under the Rural Development Programme 2007-2013 [Regulation (EC) No 1974/2006]. This indicator is composed of the population trends of bird species characteristic of farmland, intended to represent the influence of the manifold types of agricultural land-use of agriculture as well as reflecting overall biodiversity (multi-species indicator; Directorate General for Agriculture and Rural Development 2006). The purpose of this kind of indicator is to generalise complex information and thus provide simple, immediate information to policy-makers, decision-makers and the general public ("headline indicator"; Gregory *et al*. 2008).

The species trends used in the Farmland Bird Index are generally based on data collected by volunteer counters, who take part in national or regional common bird monitoring schemes run by non-governmental organisations (for an overview of schemes see http://www.ebcc.info). In Austria, the data for the population trends comes from the Austrian Breeding Bird Monitoring scheme ("Monitoring der Brutvögel Österreichs"), which was started in 1998 and has since then been organised by BirdLife Austria (e.g. Teufelbauer 2010b).

The aim of this study was to select a tailor-made basket of indicator species for the Austrian Farmland Bird Index. The EU has proposed a non-obligatory list of species for the indicator, based on a European selection made originally by the EBCC (referred to as 'EU-list' hereafter; Directorate General for Agriculture and Rural Development 2006). The EBCC itself has later produced another species classification according to three broad habitat categories ´common farmland species´, ´common forest species´, and ´other common species´. These species sets were differentiated for four European sub-regions. In this approach Austria lies in the ‘Continental’ sub-region which covers Central and East Europe: Austria, Czech Republic, East Germany, West Germany (continental part), France (continental part), Hungary, Italy (continental part), Poland, Switzerland, as well as potentially Romania, Bulgaria, Slovakia, and Lithuania. The other regions are ´Atlantic´ (West Europe), ´Mediterranean´ (South Europe), and ´Boreal´ (North Europe; Voříšek & Klvaňová 2010). This classification was updated in 2007 (and is referred to as ‘EBCC 2007-list’ hereafter).

For the application in Austria we did not consider any of these selections to be the best choice, because (a) from an Austrian point of view not all of the proposed indicator species qualify as farmland birds in today’s agriculture, (b) some species are rare as breeders or even extinct and therefore cannot be monitored using the standardized count methods for common birds, and (c) a few species we regard typical of Austrian farmland were not considered in either supranational selection. This is especially true for birds associated with mountain farming, which is very important in Austria and includes large areas of alpine pastures (>700 000 hectares). A total of 101 265 farms with an area of 1.75 million hectares of land under agricultural use qualify as Less Favoured Mountain Areas according to the EU’s criteria (54.1% of all farms and 54.7% of the overall area under agricultural land use, respectively; BMLFUW 2009).

Material and Methods

The Austrian Breeding Bird Monitoring

The Austrian Breeding Bird Monitoring uses point counts. Each count site comprises 10-15 points (12,1 ± 3.3 mean ± SD). In open landscapes the minimum distance between two points should not fall below 400 metres. The locations of the majority of the points are selected by the observers. Points are visited twice in the breeding season. For each visit a time frame of about 15 days is scheduled (encompassing three weekends). Between 1998 and 2008, visits took place on average between April 15th and April 29th, and between May 20th and June 4th, respectively. Minor deviations from the given time frames are tolerated.

At each visit, birds are counted for five minutes per point, after a short waiting period to account for possible disturbance caused by the approaching observer. All birds seen or heard are noted in three categories (“singing”, “calling”, and “visual observation”). Certain double counts are not recorded and uncertain identifications are discarded. Counts take place only under good weather conditions. Habitat data is recorded by the counters according to a simple two-level system, in a circle centred at the count point (radius between 50 and 200 m, depending on the habitat; Dvorak & Teufelbauer 2008).

To allow comparability over the years, (1) a site must be counted by the same person(s), (2) the points of a site have to be counted in the same order at each visit and (3) the counts have to take place at roughly the same time of the day. Most of the counts take place in the early morning. Until 2008 the counts were restricted to altitudes below 1200 m. An extensive description of the counting method is given by Dvorak & Teufelbauer (2008). Between 1998 and 2008, roughly 140 volunteer counters collected data at some 175 sites every year.

Population trends were calculated using the standard procedure suggested by van Strien & Soldaat (2008), using TRIM software (Pannekoek & van Strien 2001). Trends for each indicator species are stratified according to population size in the nine Austrian federal states (post-hoc stratification). For details see Teufelbauer (2010b).

Species	**Austria**	**DG Agri**	**‚EBCC 2007'**
Ciconia ciconia			
Falco tinnunculus	**x**	**x**	**x**
Perdix perdix	**x**		**x**
*Burhinus oedicnemus***		x	
Vanellus vanellus	**x**	**x**	**x**
Limosa limosa		x	
Columba palumbus		**x**	
Streptopelia turtur	**x**	**x**	**x**
Jynx torquilla	**x**		
*Melanocorypha calandra**			x
*Calandrella brachydactyla**			x
Lullula arborea	x		
Alauda arvensis	**x**	**x**	**x**
Galerida cristata		x	x
Hirundo rustica		**x**	**x**
Anthus trivialis	**x**		
Anthus pratensis			x
Anthus spinoletta	x		
Motacilla flava		x	x
Saxicola rubetra	**x**	**x**	**x**
Saxicola torquata	**x**		**x**
Oenanthe oenanthe	x		
Turdus pilaris	**x**		
Acrocephalus palustris	**x**		
Sylvia communis	**x**	**x**	**x**
Lanius collurio	**x**	**x**	**x**
*Lanius senator**		x	x
Corvus frugilegus			x
Sturnus vulgaris	**x**	**x**	**x**
Passer montanus	**x**	**x**	**x**
Serinus serinus	**x**		
Serinus citrinella	x		
Carduelis carduelis	**x**	**x**	
Carduelis cannabina	**x**		**x**
Emberiza citrinella	**x**	**x**	**x**
*Emberiza cirlus***			x
*Emberiza hortulana***			x
Miliaria calandra	**x**	**x**	**x**
species numbers			
total	24	19	24
trend data available	20	14	15

Table 1. Indicator species for the Austrian Farmland Bird Index compared to the 'EU-list' and the 'EBCC 2007-list' for continental Europe (Voříšek & Klvaňová 2010). A bold '**x**' indicates that trend data for this species is available in the period 1998-2008. * species does not breed in Austria; **less than 25 breeding pairs in Austria

Indicator species selection

Species selection started with a long list of potential indicator species. This list included all species of the 'EU-list', all species labelled as 'farmland bird' according to the 'EBCC 2007-list', as well as all species recorded mainly on agricultural land in the Austrian Breeding Bird Monitoring (see Table 2 and below). We added farmland species currently not recorded sufficiently because of missing counts in higher altitudes (e.g. Water Pipit *Anthus spinoletta*), as well as some birds, for which the connection to agriculture should be examined in detail. We did not add any rare farmland bird species, for which sufficient sample sizes cannot be obtained via point counts in Austria (e.g. Curlew *Numenius arquata*).

Subsequently, the suitability of all candidate indicator species was evaluated based on the following points:

Strong linkage to farmland and farming practices:
This criterion is to ensure that the Austrian populations of the species in question are exposed to as few factors as possible other than those of agricultural origin. First, we used the habitat association data collected during the counts of the Austrian Breeding Bird Monitoring – a previous study showed that the count points of our scheme give a highly representative picture of the farmed landscape for altitudes below 1 200 m (Frühauf & Teufelbauer 2006). Candidate species needed to have more than 50 % of records on count points with >50 % agricultural land-use. Species with strong links to other habitats, e. g. settlements (Barn Swallow *Hirundo rustica*, House Martin *Delichon urbica*) or woodland (Stock Dove *Columba oenas*, Ring Ouzel *Turdus torquatus*) were excluded.
In addition to the habitat data from the ABBM counts, we used a variety of sources for the assessment: the results of studies on links between farmland birds, habitat and land-management (Frühauf 2004, Frühauf & Teufelbauer 2006), modelled distributions of the species analysed in a study on the assignment of the Austrian High Nature Value Farmland (J. Frühauf, unpublished), published comprehensive information on ecology (Glutz von Blotzheim et al. 1966-1997, Bauer et al. 2005) and Austrian population estimates linked to the habitat classes of CORINE Landcover 1990 (BirdLife Austria, unpublished).

Reliable recording:
Using the count method of the Austrian Breeding Bird Monitoring scheme. For example, this criterion excludes nocturnal species and colonial breeders.

Frequency of occurrence:
According to our experience with trend calculation, we set 30 sites per species per year as the preferable minimum sample size. In the case of species falling below this minimum, yet qualifying as an indicator species according to the other points, this means that either its sample size has to be increased in the near future to meet this criterion, or that this species has to be excluded (e.g. Citril Finch *Serinus citrinella*, Hoopoe *Upupa epops*).

Reliable breeding status of counted birds:
Species with complex breeding biology (Quail *Coturnix coturnix*) or unclear breeding status during the count periods (birds on migration vs. breeding birds; Marsh Harrier *Circus aeruginosus*) were also excluded.

Lack of persecution and exploitation by humans:
Species which are persecuted by humans (partly illegally) to a substantial extent were excluded (e.g. Common Buzzard *Buteo buteo*, Carrion Crow *Corvus corone*, Magpie *Pica pica*). As were species for hunting purposes (Pheasant *Phasianus colchicus*).

Note that a distribution covering the majority of Austria was no prerequisite for selection. Some typical farmland birds occur almost exclusively in eastern Austria, where bird diversity and proportion of farmland is highest (e.g. Turtle Dove *Streptopelia turtur*, Corn Bunting *Miliaria calandra*), whereas other species are mainly restricted to the western alpine part of the country (e.g. Northern Wheatear *Oenanthe oenanthe*). All of these species are substantial components of overall Austrian farmland biodiversity and hence have been kept as candidate indicator species.

Results

The Austrian species selection for the Farmland Bird Index resulted in 24 indicator species which met the applied criteria (Table 1). Table 2 gives an overview of all species originally considered and highlights crucial points standing against selection. Species with a strong argument against selection were excluded from the indicator list, but species with only a weak argument against selection were retained. The Austrian species basket includes 12 (50 %) of the species that were also proposed indicator species in the 'EU-list', and 14 species (58 %) which are labelled as farmland bird in the 'EBCC 2007-list' for continental Europe (Table 1).

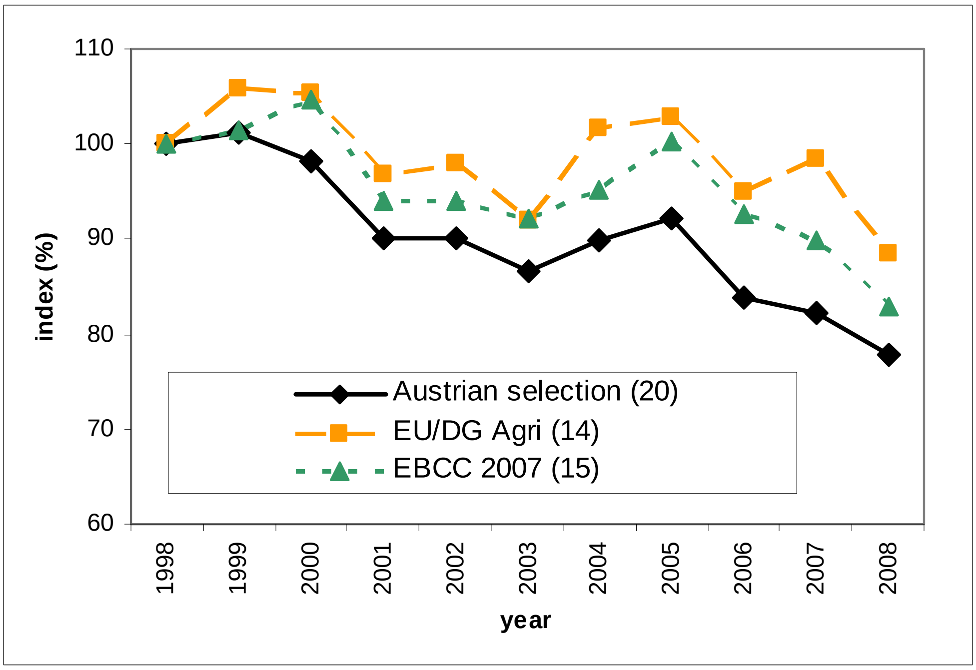

Figure 1. Farmland Bird Index for Austria: Austrian species selection compared to the 'EU-list' and to the 'EBCC 2007-list' for Continental Europe (Voříšek & Klvaňová 2010). See Tab. 1 for details. The number of species used for the indicator is given in brackets. This indicator does not currently include any data from mountain farming, above 1200m.

Species selection	Linear Regression	
	slope	r^2
Austria	-2,05	0.84
EU-list	-0,93	0.33
EBCC 2007-list	-1,4	0.57

Table 3. Results of linear regressions the three indicator trends of Figure 1.

So far, trend calculation is possible for 20 out of the 24 Austrian indicator species although some trends are based on sample sizes below the proposed threshold (see sample sizes in Table 2). The Farmland Bird Index resulting from these trends – which covers only lowland farmland – is shown in Figure 1, which also depicts the trend of the multi-species indicators derived from the species selections of the 'EU-list and the 'EBCC 207-list'. The Austrian selection results in a more pronounced negative trend than both other sets of species (Table 3). However, the trends of all three indicators are correlated significantly to each other (Austria to EU-list, r_s =0.782, p=0.007; Austria to EBCC-2007-list, r_s=0.918, p<0.0001; EU-list to EBCC-2007-list, r_s =0.891, p=0.0004).

Discussion

The final Austrian species basket differs from both the 'EU-list' and the 'EBCC 2007-list'. Whereas the degree of overlap between the Austrian and both supranational species selections seems to be poor at first sight, it should be noted that, if only species with sample sizes sufficiently large for trend calculation in Austria are counted, only two ('EU-list'; Woodpigeon, Barn Swallow) respectively one ('EBCC 2007-list'; Barn Swallow) of the species proposed on a supranational level are not considered in the Austrian species selection. All other species have to be ruled out because they are very rare breeding birds, extinct or have never bred in Austria (see Ranner 2010). Consequently, the major difference between the Austrian and the two other lists of indicator species lies in the additional species of the Austrian species basket. Given the habitat linkages of the Austrian populations of the selected species we think that their use as indicators for farmland is justified, and that our indicator gives a much more representative picture of Austria's farmed landscape's biodiversity. In particular, mountain farming in Austria should now be represented in the indicator by species such as Water Pipit *Anthus spinoletta* and Northern Wheatear *Oenanthe oenanthe,* as well as by (sub)alpine populations of Skylark *Alauda arvensis*, Tree Pipit *Anthus trivialis*, Whinchat *Saxicola rubetra*, and Linnet *Acanthis cannabina*.

However, for some of the Austrian indicator species it is necessary to increase the sample size to obtain reliable trends (see Table 1). This is especially true for species associated with high altitude farming, for which almost no data in the common bird monitoring scheme is available for the first ten years. An extension of the monitoring scheme has been implemented from 2008 onwards (Frühauf & Teufelbauer 2008). Because the number of volunteer counters in Austria is not large enough to cover the required number of count sites, the use of professional counters for alpine common bird monitoring sites has been established since 2008, with funding from the Austrian Ministry for Agriculture. With this extra input, achieving the minimum sample size for most of the indicator species seems feasible, and based on recent data, only the sample size for Citril Finch – a localised breeding bird in the western part of alpine Austria – will probably remain below the threshold for trend calculation (Teufelbauer 2010a).

Naturally, the species selection influences the trend of the derived indicator. The 'EU-list' results in the least negative trend, whereas the Austrian-targeted selection results in the most pronounced decrease over time (Figure 1). Given that the Austrian species basket is specially designed to depict the national situation, rather than the more broader continental view provided by the 'EU-list' or 'EBCC 2007-list', its use has to be preferred over both other species selections, regardless whether its trend matches them or not. Moreover, the use of the less country-specific indicators of the supranational species selections could be misleading, because both draw a less negative picture of the fortunes of Austria's farmland birds. The more negative trend of the Austrian species selection compared to the two European lists might also indicate a higher sensitivity of the Austrian indicator. Given a likely aim of the indicator – to act as an early warning system for the state of biodiversity – a more sensitive indicator is clearly advantageous. Possible impacts of of agri-environmental measures should therefore also be detected more readily, and earlier, by the Austrian species selection. Lastly, an indicator which takes into account mountain farming systems better reflects the importance of Austria for bird populations dependent on those habitats in the European Union.

In recent work, the Austrian Farmland Bird Index 1998-2008 has been found to be stable in relation to changes in species, the slope of the indicator changing little when one or several of the indicator species are omitted by chance, as shown Teufelbauer 2010b. This result strengthens the credibility of the Austrian Farmland Bird Index as a stable indicator system.

Acknowledgements

The development of the Austrian Farmland Bird Index was funded by the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (project BMLFUW-LE.1.3.7/0005-II/5/2007). Major contributions to this work have been made by Georg Bieringer, who also made valuable comments on a draft of this paper, Michael Dvorak, and Gábor Wichmann. Valuable comments on the species selection were made by Josef Feldner, Armin Landmann, Katharina Peer and Martin Pollheimer. Sincere thanks go to Benjamin Seaman for improving the English.

This work would not have been possible without the invaluable work of many volunteer counters, who have collected an enormous quantity of data that builds the backbone for the calculation of population trends and the Farmland Bird Index in Austria.

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MS received: 16.05.2010 Revised MS accepted: 30.03.2011

Table 2. Summary of data on candidate species for the Austrian Farmland Bird Index. 'Crucial points' highlights points of concern in the discussion on the suitability as indicator species. ABBM Austrian Breeding Bird Monitoring, • strong argument against selection and hence not selected as an indicator species, (•) weak argument against selection and retained as an indicator species.

Indicator species	**Species name**	**Breeding population**[1]	**% of population in habitat class**[2]					**ABBM (98-06 mean)**			**Crucial points**				
		Pairs (mean)	**Farmland**	**Woodland**	**Urban / artificial surfaces**	**Water bodies**	**Not attributable**	**No. of Sites**	**No. of birds**	**% of birds on farmland**[3]	**Habitat**	**Reliable recording**	**Frequency / distribution**	**Biology**	**Persecution**
yes	*Falco tinnunculus*	7 500	59,5	2	18	0	20,5	82	285	88					(•)
yes	*Perdix perdix*	9 000	94	0	1	0	5	23	82	98			(•)		(•)
yes	*Vanellus vanellus*	4 500	n.a.					34	296	98					
yes	*Streptopelia turtur*	11 500	n.a.					50	228	53	(•)				
yes	*Jynx torquilla*	3 500	n.a.					18	33	55	(•)		(•)		
yes	*Lullula arborea*	800	n.a.					6	35	95			(•)		
yes	*Alauda arvensis*	180 000	94	0	0	0	6	68	1 646	98					
yes	*Anthus trivialis*	52 500	80	18	0	0	2	43	111	61					
yes	*Anthus spinoletta*	90 000	57	7	0	0	36	2	13	61					
yes	*Saxicola rubetra*	5 250	n.a.					26	133	86					
yes	*Saxicola torquata*	6 750	n.a.					30	91	95					
yes	*Oenanthe oenanthe*	6 750	n.a.					11	31	90					
yes	*Turdus pilaris*	45 000	66	30	3	0	1	38	257	67	(•)				
yes	*Acrocephalus palustris*	30 000	n.a.					54	209	85					
yes	*Sylvia communis*	30 000	86	0	7	0	7	40	129	87	(•)				
yes	*Lanius collurio*	30 000	91	5	1	0	3	62	177	88					
yes	*Sturnus vulgaris*	150 000	32	44	23	0	1	124	3 527	84	(•)	(•)			
yes	*Passer montanus*	120 000	58	2	36	0	4	85	836	88	(•)				
yes	*Serinus serinus*	67 500	72	0	27	0	1	59	225	81	(•)				
yes	*Serinus citrinella*	3 250	n.a.					2	4	0			(•)		
yes	*Carduelis carduelis*	37 500	81,5	4	12.5	0	2	72	273	69					
yes	*Carduelis cannabina*	18 000	87	0	11	0	2	27	201	92					
yes	*Emberiza citrinella*	90 000	91	6	0	0	3	122	1 076	72					

Indicator species	Species name	Breeding population[1]	% of population in habitat class[2]					ABBM (98-06 mean)			Crucial points				
		Pairs (mean)	Farmland	Woodland	Urban / artificial surfaces	Water bodies	Not attributable	No. of Sites	No. of birds	% of birds on farmland[3]	Habitat	Reliable recording	Frequency / distribution	Biology	Persecution
yes	*Miliaria calandra*	5 250	90	0	4	0	6	19	94	94					
No	*Ciconia ciconia*	390	n.a.					8	23	93	•		(•)	•	
No	*Circus aeruginosus*	350	n.a.					18	48	94	•			•	•
No	*Buteo buteo*	10 000	51.5	46.5	0	0	2	99	255	68	(•)				•
No	*Coturnix coturnix*	10 000	95	0	0	0	5	32	82	96				•	
No	*Phasianus colchicus*	?	n.a.					98	1 112	76					•
No	*Burhinus oedicnemus*	12	n.a.					0	0		•	•	•		
No	*Limosa limosa*	84	n.a.					1	3		•	(•)	•		
No	*Columba oenas*	3 500	n.a.					31	109	49	•				
No	*Columba palumbus*	30 000	46	51	1	0	2	139	963	53	•				
No	*Upupa epops*	500	n.a.					9	18	75			•		
No	*Picus viridis*	10 500	n.a.					76	167	55	•				
No	*Dendrocopos syriacus*	3 000	n.a.					7	11	44	•				
No	*Melanocorypha calandra*	0	n.a.					0	0				•		
No	*Calandrella brachydactyla*	0	n.a.					0	0				•		
No	*Galerida cristata*	425	n.a.					3	8		•		•		
No	*Hirundo rustica*	130 000	74	0	24	0	2	99	831	81	•				
No	*Delichon urbica*	75 000	n.a.					47	329	70	•				
No	*Anthus pratensis*	1 800	n.a.					6	9				•		
No	*Motacilla flava*	550	n.a.					6	21	94	(•)		•		
No	*Motacilla alba*	75 000	55	0	44	0	1	95	488	81	•				
No	*Prunella collaris*	12 000	38	0	0	0	62	0	0	0	•				
No	*Luscinia megarhynchos*	3 000	n.a.					24	134	83	•				
No	*Phoenicurus phoenicurus*	9 000	n.a.					27	60	64	•				
No	*Turdus torquatus*	75 000	15	85	0	0	0	10	26	67	•				
No	*Turdus viscivorus*	100 000	5	95	0	0	0	67	271	40	•				

Indicator species	**Species name**	**Breeding population**[1]	**% of population in habitat class**[2]					**ABBM (98-06 mean)**			**Crucial points**				
		Pairs (mean)	**Farmland**	**Woodland**	**Urban / artificial surfaces**	**Water bodies**	**Not attributable**	**No. of Sites**	**No. of birds**	**% of birds on farmland**[3]	**Habitat**	**Reliable recording**	**Frequency / distribution**	**Biology**	**Persecution**
No	*Locustella naevia*	1 850	n.a.					10	20	76	(•)		•		
No	*Acrocephalus schoenobaenus*	10 000	n.a.					4	10	0	(•)		•		
No	*Sylvia nisoria*	1 550	n.a.					11	27	89		•	(•)	(•)	
No	*Sylvia curruca*	52 500	60	33	5.5	0	1,5	30	49	68	•				
No	*Lanius senator*	0	n.a.					0	0				•		
No	*Pica pica*	12 000	95	0	2	0	3	57	240	86	(•)				(•)
No	*Corvus monedula*	4 100	n.a.					23	204	69	•	•			(•)
No	*Corvus frugilegus*	900	n.a.					6	83	98	•	•			•
No	*Corvus corone*	60 000	56.5	20	20	0	3,5	155	3 132	76	(•)				•
No	*Passer domesticus*	525 000	11.5	0	88.5	0	0	67	802	81	•				
No	*Carduelis chloris*	165 000	59.5	0	38.5	0	2	132	782	70	•				
No	*Carduelis flammea*	22 500	42	52	6	0	0	2	9	27	•				
no	*Emberiza cirlus*	8	n.a.					1	1				•		
no	*Emberiza hortulana*	20	n.a.					2	6				•		
no	*Emberiza schoeniclus*	9 500	n.a.					12	23	71	(•)		•		

1 Source: BirdLife Austria, unpublished

2 combined classes of CORINE 1990

3 circles around count points with farmed area >50%

Bird Census News 2010, 23/1-2: 99-103

Bird monitoring on an urban scale: the case of Brussels

Anne Weiserbs & Jean-Yves Paquet

Abstract. A special environmental scheme is dedicated to the autonomous Region of Brussels. The official bird monitoring started in 1992 thanks to the support of volunteers. The different modules composing this monitoring are presented..

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Introduction

The city of Brussels (169 km²) is an autonomous region in the federal state of Belgium and, as such, has organised specific environmental monitoring since 1992. The bird monitoring consists of several complementary survey, based on yearly or multi-yearly field samplings. One of the most important recent outcomes of the breeding bird monitoring schemes in Brussels has been the publication of an atlas of Brussels breeding birds in 2000-2004 (Weiserbs & Jacob 2007). Here we present the basic modules of the monitoring.

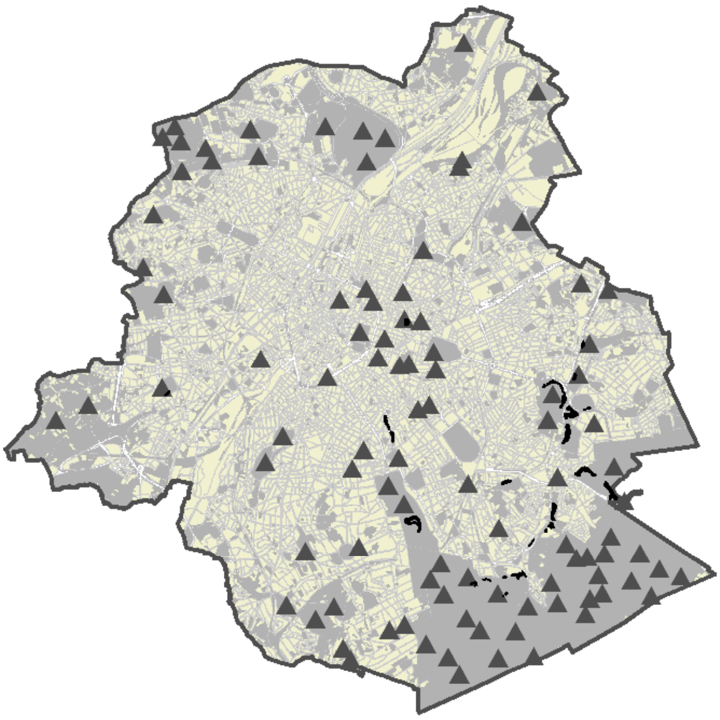

Figure 1. Location the 99 point count plots (black triangles) of the common breeding bird monitoring and the 20 ponds of the waterbid scheme (in black) in the Region of Brussels. Green area in grey.

The Brussels common breeding bird monitoring

The common breeding bird monitoring is the most important survey of this scheme. It involves 99 point counts sampled twice a year. The plots are mostly located in the green spaces of the town (Figure 1). Trends are calculated yearly using TRIM (Pannekoek & van Strien 2010). In the 1992-2009 period, 15 species were in progression, 12 were in decline and 9 were stable (Figure 2). This apparently well-balanced outcome only concerns the part of the Brussels avifauna having the most widespread species.

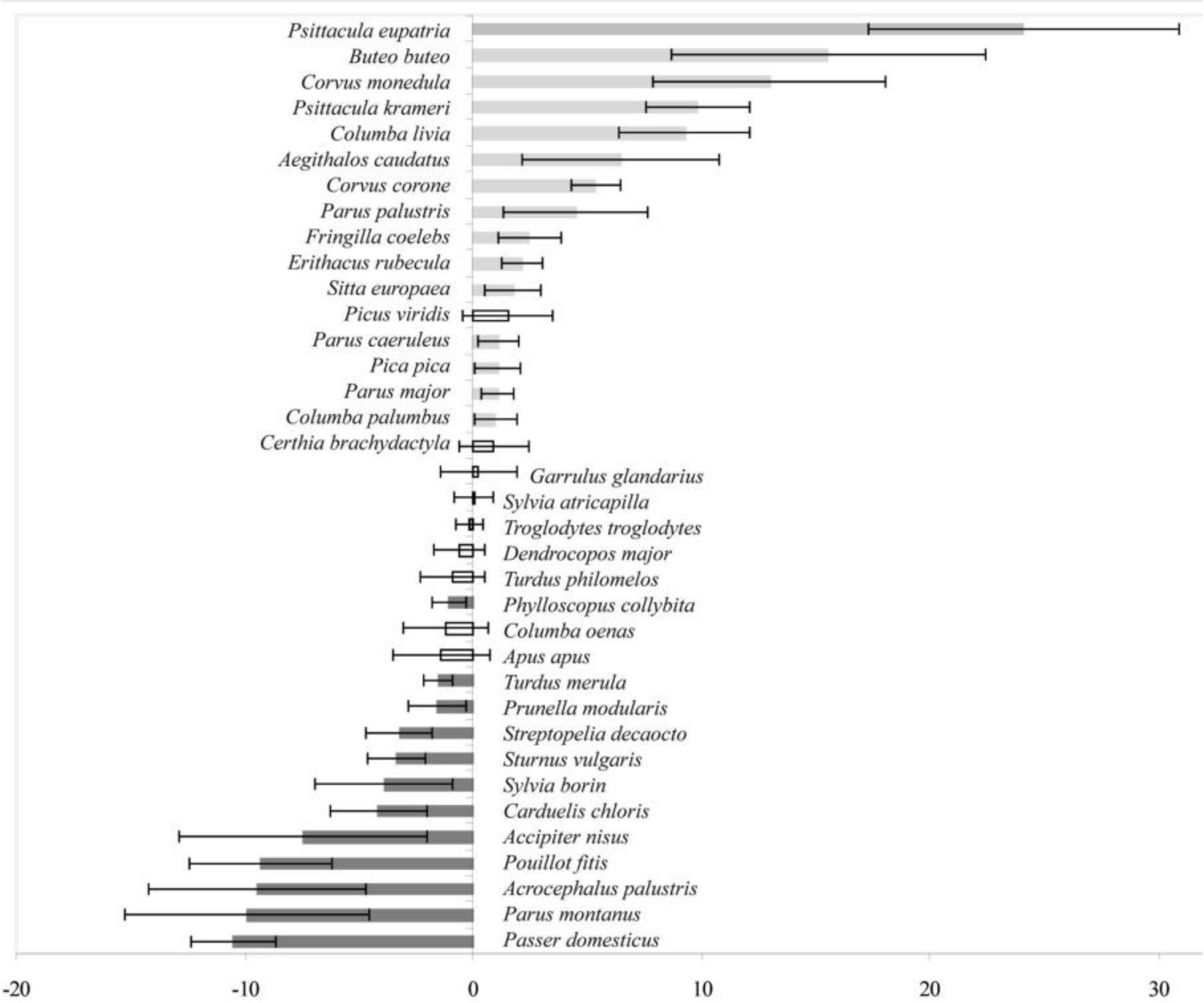

Figure 2. Common breeding bird monitoring: results from the 1992-2009 period.
Light grey: increasing species, white: stable, dark grey: increasing.

When the analysis suggests a change in the trend, focuses are made on different periods. For example, the Short-toed Treecreeper *Certhia brachydactyla* (Figure 3) increased in 1992-2004 (4 %/year, p<0,01) and steeply declined in 2004-2009 (-14%/year, p<0,01). Geographic focuses are also done, especially comparison between the trends in the Soignes forest (1 600 hectares in the South part of Brussels) and the rest of the Region. When significantly different, the situation is most often better in the town than in the forest (for example in 1992-2008, among the 12 species concerned, 8 had a better evolution in the town - Weiserbs, 2008).

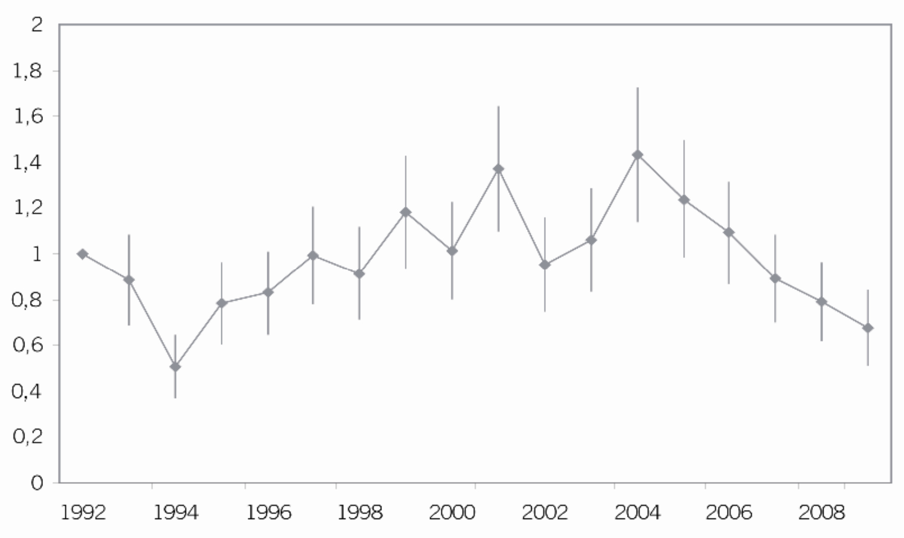

Figure 3. Evolution of the Short-toed Treecreeper *Certhia brachydactyla* in 1992-2009.

Another application is the producing of trend maps based on regressions on each point count and using the technique of Inverse Distance Weighted to interpolate these values. Figure 4 shows the example of the House Sparrow *Passer domesticus*, whose population was estimated at 2.100-4.200 breeding pairs in 2000-2004 (Weiserbs & Jacob 2007) and underwent a steep decline in 1992-2009 (-10,47 %/year, p<0,01). The trend map suggests a strong decline in the centre of the town, while the species situation seems healthy in some neighbouring areas.

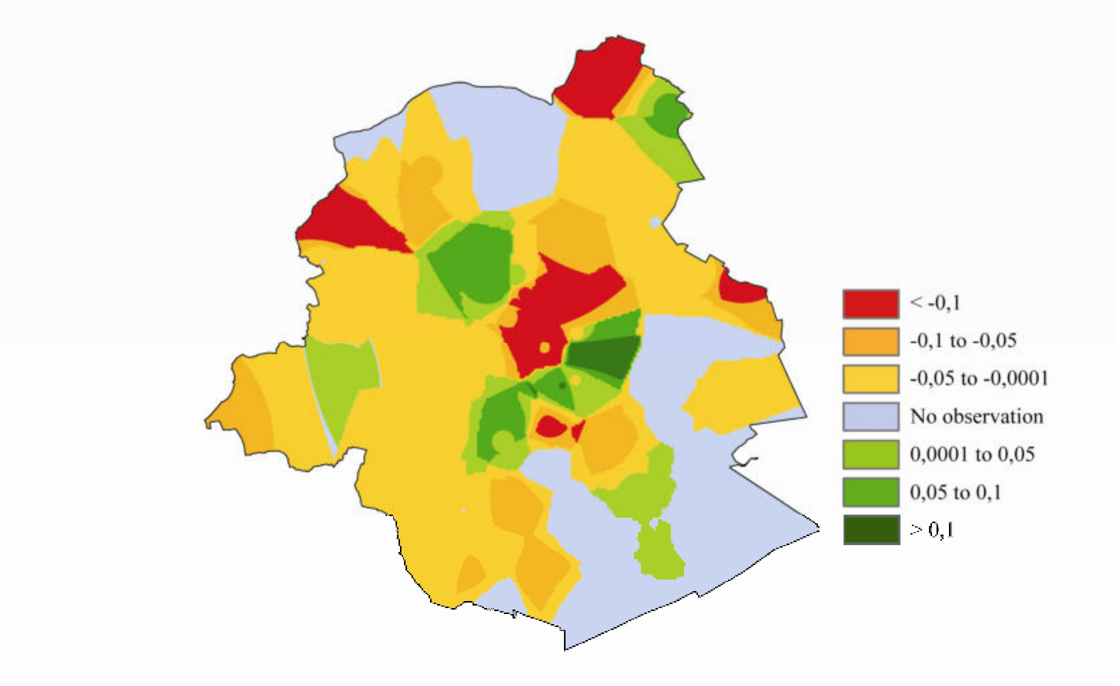

Figure 4. Trend map of the House Sparrow *Passer domesticus* in 1992-2009 based on slope of regression on each point count

The common breeding waterbird scheme

The monitoring of common breeding waterbirds consists of a yearly spring census in the 20 major wetlands and ponds of the Region (see Figure 4). During the 1995-2009 period, the population of 3 indigenous species remained stable (Coot *Fulica atra*, Moorhen *Gallinula chloropus* and Mallard *Anas platyrhynchos*), 2 showed a steady increase (Tufted Duck *Aythya fuligula* 12 %/year and Mute Swan *Cygnus olor* 7 %/year) and one decreased (Great Crested Grebe *Podiceps cristatus* 6 %/year). In the same period, 2 exotic species increased sharply: Canada Goose *Branta canadensis* (50 %/year) and Egyptian Goose *Alopochen aegyptiacus* (8 %/year) and one seemed stable (Mandarin Duck *Aix galericulata*).

Specific monitoring schemes

Some species or group of species benefit from specific monitoring schemes. This is the case of the House Martin *Delichon urbicum* (Figure 5) and of two breeding Parrots, the Ring-necked Parakeet *Psittacula krameri* and the Alexandrine Parakeet *P. eupatria*, sharing two roosts whose summer numbers exceeded 9 400 birds in 2009 (mostly Ring-necked Parakeet, Figure 6).

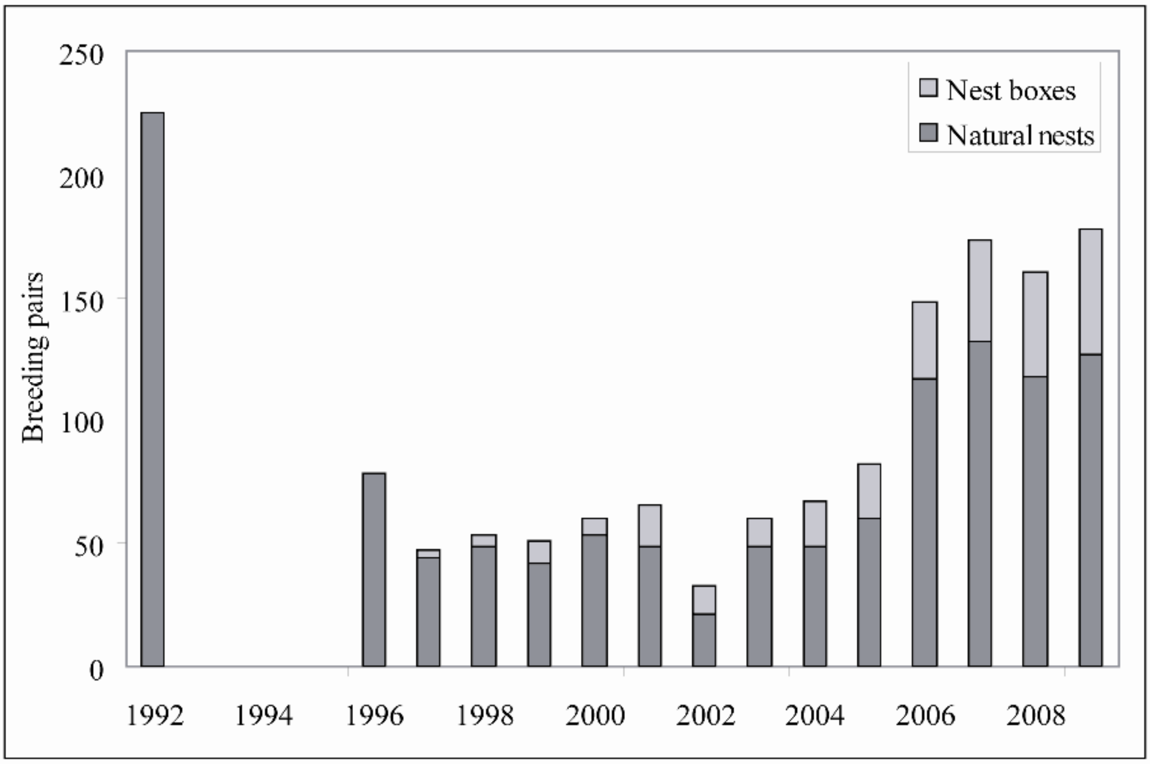

Figure 5. Result of the specific scheme for the House Martin, *Delichon urbicum*

In 2010, two new schemes started: one in the Soignes forest for the Woodcock *Scolopax rusticola* and the Long-eared Owl *Asio otus*, and the other to make an inventory of the remaining breeding pairs of the Barn Swallow *Hirundo rustica*. From 2011, a scheme for both Black Woodpecker *Dryocopus martius* and Middle Spotted Woodpecker *Dendrocopos medius* is planned.

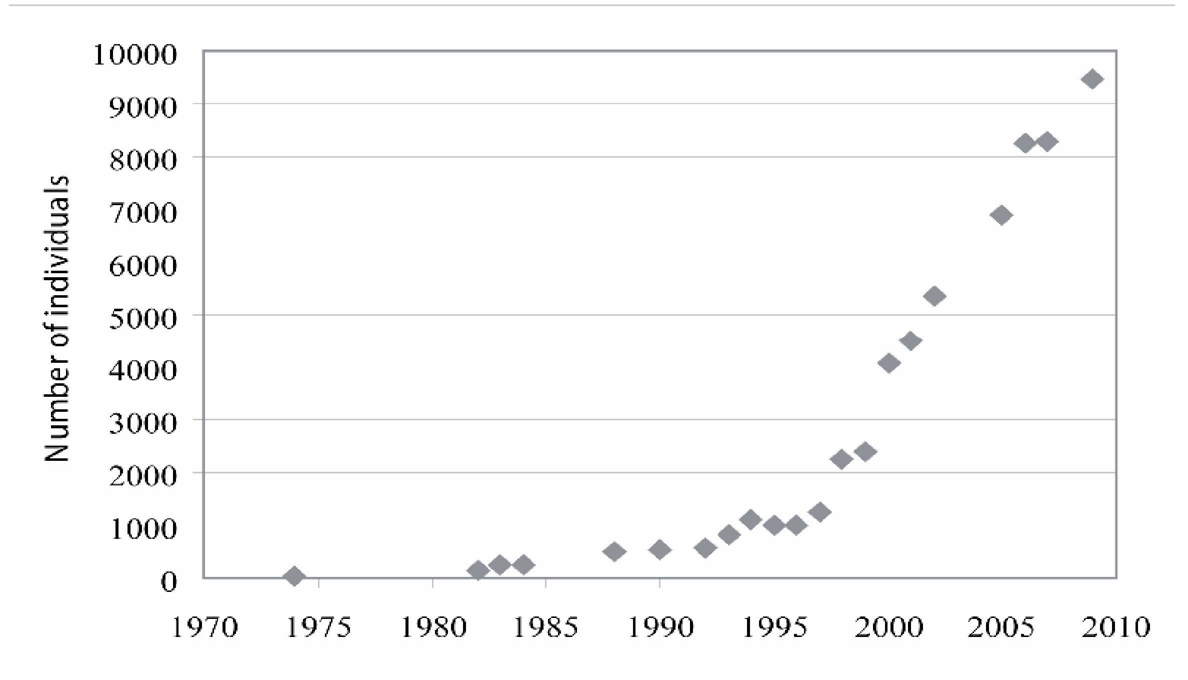

Figure 6. Summer counts at roosts shared by the Ring-necked Parakeet *Psittacula krameri* and the Alexandrine Parakeet *P. eupatria*

Conclusion

In a complementary way the different modules contribute to the avifauna monitoring of Brussels, giving field information to the authorities not only to support conservation policy, but also to give practical advice for green space management. These modules involve about half of the regional breeding species. The rest of the avifauna, mostly rare species as 46 species on the 103 regional breeders have a population of less than 20 pairs (Weiserbs & Jacob 2007), is monitored through targeted inventories like atlases.

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MS received: 15.06.2010
Revised MS accepted: 02.10.2010

Bird Census News 2010, 23/1-2: 105-112

Meadow bird monitoring in France and Russia : first results of comparative researches

Joël Broyer[1] , Alexander Mischenko[2], Olga Sukhanova[2] & Laurence Curtet[1]

Abstract. National monitoring of meadow ecosystem has been implemented in France since 2001 and in Russia since 2007 to describe meadow bird annual trends (France) or trends across successive periods of 3 years (Russia). One thousand 12-ha sampling plots from > 100 different regions are studied in France, 192 sampling plots from 36 study regions in Russia. A common index was defined corresponding to the number of individual passerines recorded during two 15-minute sessions by a static observer. The results obtained in France and Russia seem to indicate that passerine abundance is a function of the timing of mowing.

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Introduction

In western Europe, farmland birds, meadow birds in particular, have undergone a severe decline in the last decades after long term changes in farming practices (Vickery et al. 2001; Newton 2004). The EU Common Agricultural Policy has long encouraged massive conversions of grassland into arable land. In France for example, permanent grassland areas decreased at a mean rate of 108 000 ha per year across the 1989-2005 period. Moreover, early mowing, the consequence of which on species like the corncrake *Crex crex* has been described for many decades (Norris 1947; Von Haartman 1958) and the impact of management intensification on invertebrate abundance (Britschgi et al. 2006) may interfere with bird reproduction by decreasing breeding outputs (Roodbergen & Klok 2008; Schekkerman *et al*. 2009). In Russia by contrast, a long-lasting crisis in agriculture leads to the abandonment of huge grassland areas after a decrease of more than half the total livestock number since the mid 1980's (source: Ministry of Agriculture of the Russian Federation). Abandoned areas are gradually overgrown by bushes and progressively lost as habitat for meadow birds (Mischenko & Sukhanova 2006). Agri-environment schemes have been implemented in western countries to mitigate the effects of intensified practices but with few successful results reported so far for meadow birds (Kleijn *et al*. 2001; Berendse *et al*. 2004). Moreover, such programmes are absent in Russia. There is therefore an urgent need for a large scale survey on meadow birds to monitor population trends and thereby assess the global efficiency of agri-environmental policy.

National Monitoring Schemes have been launched in France since 2001 and in Russia since 2007, providing comparable standardized data, collected annually in France but, due to the huge surfaces and distances involved, by periods of three successive years in Russia. The primary aim was providing information on meadow bird response to ongoing changes in farming practices, i.e. intensification in France, balanced or not balanced by agri-environmental policy, and abandonment in Russia. The opportunity was taken to compare grassland bird communities at the south-western and northern limits of their range in Europe. We investigated in particular the *a priori* prediction that late mowing, which is the main management objective of Agri-Environment Schemes in France, might influence bird abundance at the level of sampling plots. This article describes the method and presents first results of this joint programme.

Data collection

The statistical unit for data collection is a 12 ha "sampling plot" i.e. the area included within a 200-m-radius circle in which all required information must be collected. The sampling procedure is based on "study regions" i.e. large scale agricultural landscape units which are described by a variable number of sampling plots. The representativeness of the schemes was achieved in France by a quasi-systematic cover, involving virtually each “département” (Figure 1) in which local experts were asked to localize grassland areas still harbouring significant meadow bird populations. The “départements” not included in the sampling have no important meadow bird breeding area. Only hay meadows still provide favourable breeding conditions for birds.

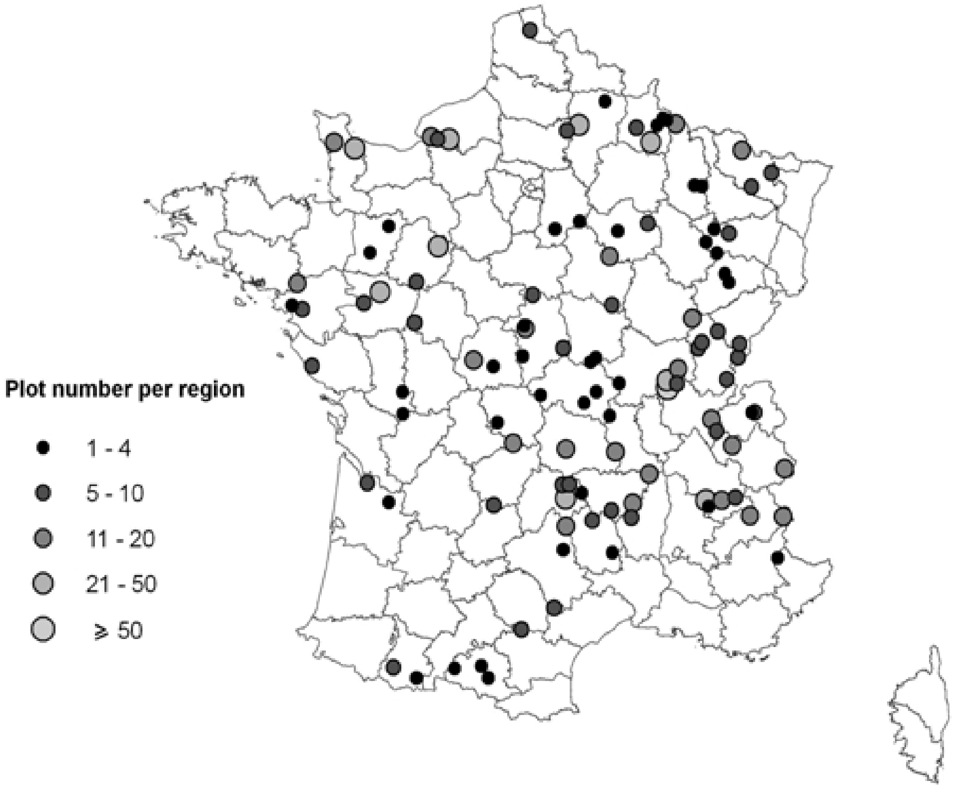

Figure 1. Study regions monitored in France by the National Hay-meadow Ecosystem Observatory (territorial subdivisions correspond to the “départments”).

In Russia, experts selected the most important "oblasts" (Figure 2) for meadow birds, in which the choice of "study regions" was stratified in 4 categories of habitat: flooded meadows, not flooded meadows, lately grazed and recently abandoned grasslands, in proportion of assumed importance for breeding birds. In both countries, sampling plot number per study region varies according to the total surface of existing habitat but also as a function of observer's available time. In each study region, plots were localised in areas *a priori* most favourable to meadow bird breeding. They were spatially stable from year to year

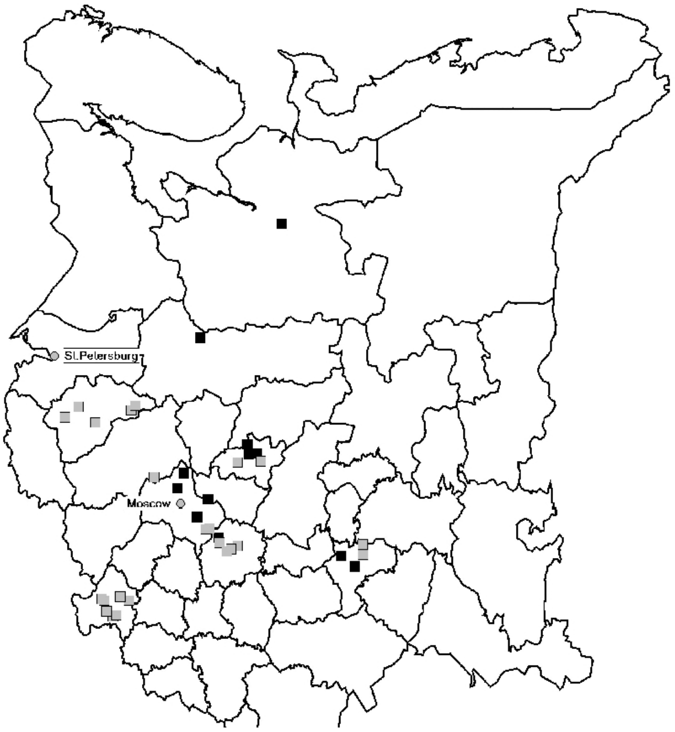

Figure 2. Monitored regions in Russia in the 2007-2009 period (territorial subdivisions correspond to the "oblasts"). Dark grey: monitored each year, light grey: monitored one year.

In France, more than 1000 sampling plots located in more than 100 study sites (Table 1) are visited annually by a network of observers from diverse environmental organizations (National Parks, Regional Parks, NGO, hunting societies, Nature Reserves,…) under the coordination of ONCFS (a governmental Agency). A bird census is performed by a static observer in the centre of each plot during two successive periods of 15 minutes, in the first and the second half of the nesting period, before juvenile fledging time and before the start of mowing . Only ground nesting species are taken into account. The point count method used here differs from those implemented for example in the French Common Bird Monitoring Scheme (Julliard & Jiguet 2005) by focalizing on a specific habitat since hay-meadow must account for at least 40% of the total area in any new plot included in the database. The percentage of total meadow area already mown is assessed annually at 4 dates: June 20, July 1, July 15 and August 1 (in upland meadows). Additional information is collected every 5 year: a description by the observers of the grass cover (height, density), of flora richness along a 10m x 2m transect, of the most abundant flora species and families; meadow management (fertilization, cutting width of mowers,…) is described by questioning farmers.

Years	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
N study regions	47	71	80	91	100	98	107	108	110	107
N sampling plots	596	763	800	874	939	958	1031	994	1026	1021

Table 1. Annual number of regions and sampling plots involved in the national monitoring programme in France

In Russia, 193 sampling plots in 36 study regions were visited either one single year (n=132) or each year (n=61), during the 2007-2009 period. Future trends will be monitored by comparing the successive results of 3-year-period surveys. Training sessions for observers were organized annually in France since 2000 and in 2006 in Russia to standardize the field work method and application. Since all data have been collected with similar method in France and in Russia, identical indices may be derived from bird counting: species richness (total number of species) and Meadow Passerine Index (MPI) i.e. meadow passerine number (sum of individuals), with for each species the maximum number recorded during the two 15-minute sessions in each sampling plot. The average MPI value may be computed for each study region.

Results

Similarity and dissimilarity between bird communities in France and in Russia

The most frequently recorded passerine species in both countries were Whinchat *Saxicola rubetra,* Skylark *Alauda arvensis,* Yellow Wagtail *Motacilla flava*, Reed Bunting *Emberiza schoeniclus* with, in France Corn Bunting *Miliaria calandra* (never recorded in Russian samples) and in Russia Whitethroat *Sylvia communis* (not recorded as open grassland breeder in French hay-meadows). However, Russian meadows differed by species richness in breeding waders (*Gallinago gallinago*, *Limosa limosa*, *Numenius arquata*, *Tringa totanus*, *Tringa stagnatilis*, *Philomachus pugnax*, *Gallinago media*) and the high frequency of the corncrake, which was the most frequently recorded non passerine species.

Compared Meadow Passerine Indices (MPI)

In France, MPI was higher in lowland meadows (≤ 200m), generally located in flooded alluvial plains, and in upland meadows (800-2100m), the lowest values being recorded at intermediate altitudes (for each year, 2007, 2008 or 2009, Kruskal-Wallis test: khi square >26, $p<0.001$) (Figure 3). In Russia, passerine abundance was higher than in most French lowland meadows (Mann-Whitney test: $Z=-9.068$, $p<0.001$) (Figure 3). A similar difference was observed between flooded and non flooded plains (for each year 2007, 2008 or 2009, $Z>4.5$, $p<0.001$) (Figure 4).

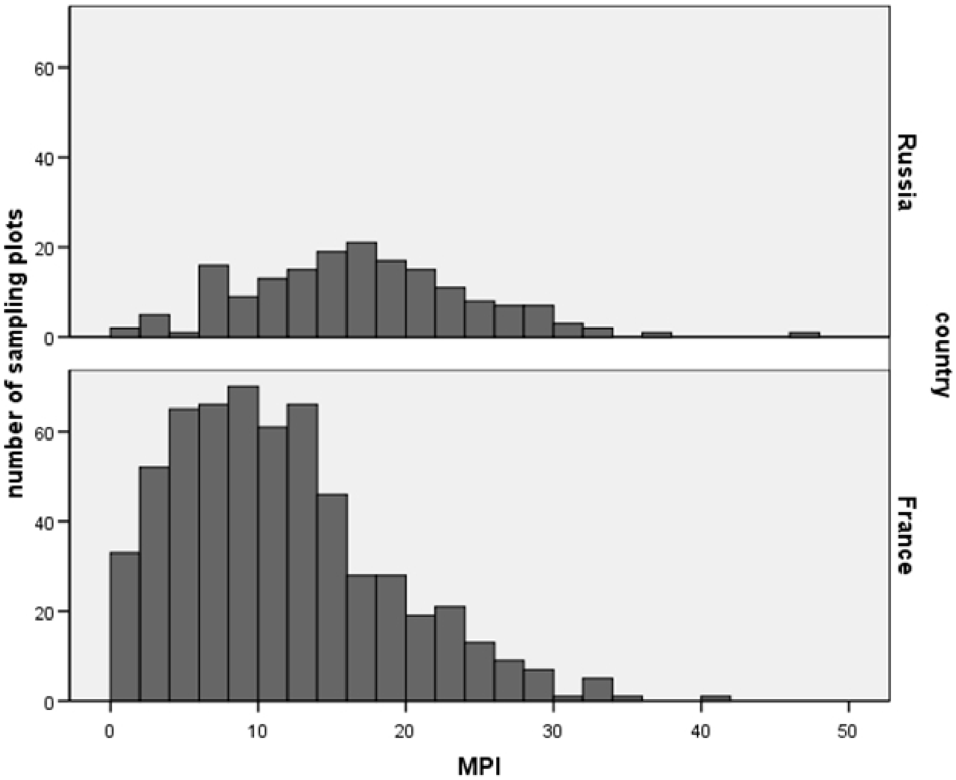

Figure 3. Meadow Passerine Indices (MPI) in Russia and in lowland French meadows mown after 1 July (2007-2008-2009).

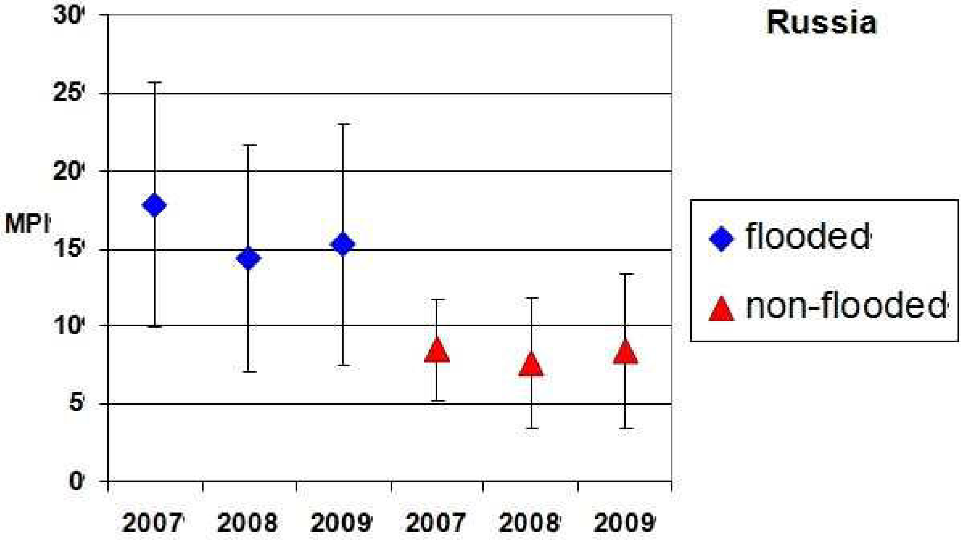

Figure 4. Meadow Passerine Index (mean and SD) in flooded and non flooded meadows in Russia in 2007, 2008 and 2009Passerine abundance and mowing schedule

In France, there is a general correlation between MPI and mowing schedule (Figure 5). More precisely, in lowland meadows, MPI in 2004, 2006, 2007 was correlated with the percentage of meadow already mown on July 1 in preceding year (2003, 2005 and 2006 respectively) (Spearman correlation rank for each year: rho>0.2, p<0.001). In 2005, MPI was correlated, not with mowing in 2004 (rho= -0.16, p=0.664), but with mowing in 2003 (two years earlier) which was influenced by a dry and hot summer (rho= -2.38, p=0.001)). In meadows (not flooded) situated at medium altitudes (200-800m), lower MPI were probably linked to earlier mowing dates. In Russia, the highest MPI values (≥ 27) were recorded in floodplain meadows mown in latest terms (after 10 July), mown incompletely or not mown in previous year.

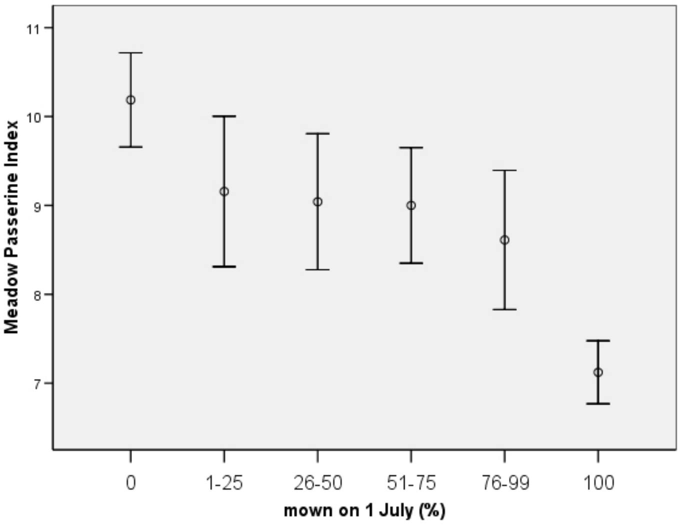

Figure 5. Relationship between mowing schedule and MPI (mean and 95% confidence interval) in French lowland meadows (2007-2008-2009).

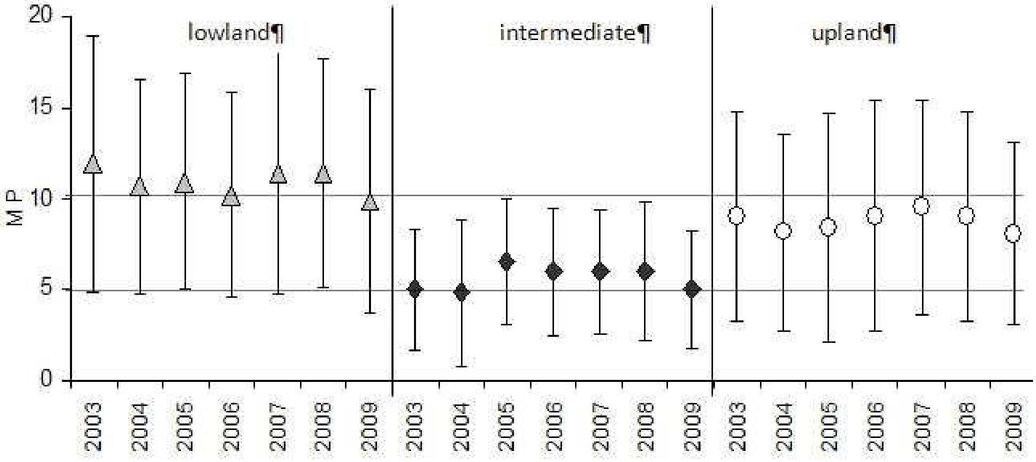

Figure 6. Meadow Passerine Index (mean and SD) in lowland (≤200m), intermediate (200-800m) and upland (800-2100m) meadows in France between 2003 and 2009.

Observed trends

In Russia, the 2007-2009 survey was the initial benchmark from which changes will be described in the future. In France, no clear annual trend in MPI was observed throughout the 2001-2009 period (Figure 6) despite the fact that mowing tended to occur earlier between 2001 and 2006 in lowland meadows. In 2007 and 2008, mowing occurred later due to rainy weather in the hay-making period and was not correlated with MPI in 2008 and 2009.

Discussion

Meadow bird breeding usually relies on traditional farming (O'Meara 1979). Changes in agricultural methods and techniques have been a continuing process for many decades. The consequences on meadow bird populations of earlier mowing in western Europe and ongoing meadow abandonment in European Russia would certainly deserve to be monitored at a relevant scale. Whereas biodiversity conservation has become a growing task in the EU agricultural policy, there is still no large scale programme allowing to assess either the impacts of farming intensification or abandonment, or the expected effects of agri-environmental schemes implemented to counterbalance these impacts. This French-Russian joint programme could be the first step towards such an European monitoring using a similar methodology that would enable to provide a global overview of how bird populations actually endure the changes in farming systems.

The first results obtained in France and in Russia seem to indicate that passerine abundance is a function of the timing of mowing. Comparison of current mowing schedules in France with meadow birds' hatching chronology (Broyer 2007) may provide convincing evidence of a cause-effect relationship. Moreover, mowing schedule was manipulated in one of the most important meadow bird breeding area in France with a clear response of MPI (Broyer 2011).

The 2007-2009 survey has also revealed the high richness of Russian meadows in non passerine birds. For example the corncrake was heard, during daytime, each year in > 70% of the sampling plots, and the Common Snipe *Gallinago gallinago* was recorded in 20-30%.

Acknowledgements

In France, the institutions members of the National Hay-meadow Ecosystem Observatory (ONEPF) involved in the data collection were: ONCFS departemental services, Vanoise, Ecrins and Pyrénées National Parks, Vercors, Bauges, Pilat, Queyras, Boucles de la Seine normande Regional Parks, Lavours, Baie de l'Aiguillon Nature Reserves, Hunting departemental Agencies of Loire and Jura, LPO 26, 38, 72, Agrenaba, CPIE vallée du Loir et la Sarthe, ANVL. In Russia, the observers were: A. Mischenko, O. Sukhanova, A. Mezhnev, S. Fokin, V. Melnikov, D. Chudnenko, V. Gridneva, O. Borodin, S. Smirnova, Y. Blokhin, O. Mezhneva, P. Zverev, D. Shitikov, D. Kuznetsov, , D. Fedchuk, V. Nikolaev, Yu. Romanov, M. Kozlova, M. Korepov, D. Korepova, M. Kazakova (botanist expert).

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MS received: 10.05.2010
Revised MS accepted: 09.02.2011

Bird Census News 2010, 23/1-2: 113-116

Can current data scheme's in Flanders assess winter impact on Grey wagtail *Motacilla cinerea* numbers ?

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Abstract. I compare six scheme's of counts in Flanders for the impact of a cold spell on Grey Wagtail numbers. Only the reporting rate in a set of 'occasional bird records' generated useful information: it dropped by 38% for the number of records and 40% for the number of birds

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Introduction

The Grey wagtail is a regular migrant in Belgium with peak autumn passage (dispersal and migration) during September-October. Despite this pattern of early migration, which suggests at least a medium-distance destination, many birds winter in western Europe, including Belgium. Grey wagtail is generally considered a winter-sensitive species in Europe, with numbers much reduced by severe winters.

Numbers of Grey wagtails increased steadily the last decades; e.g. in The Netherlands by 47% since 1990 *(Sovon 2009)*. After a long series of (record) mild winters since 1997, there were three weeks of severe frost over large parts of western and south-western Europe in January 2009.

In this paper, I investigate if the current bird monitoring scheme's in Flanders were able to asses the impact (if any) of this cold spell in winter on Grey wagtail numbers.

Material and methods

Grey wagtail is included in the following bird count schemes in Flanders:

(1) common breeding bird monitoring (ABV),
(2) monitoring of rare and uncommon breeding birds (BBV),
(3) winter bird counts (PTT),
(4) migration counts (www.trektellen.nl),
(5) bird ringing,
(6) 'occasional' records reported to www.waarnemingen.be, an online data portal capturing about a million bird records per year.
We calculated the 'reporting rate' as the species' statistic, i.e. the proportion of records of Grey wagtail amongst all bird records reported.

Results

In the common breeding bird monitoring (ABV) in Flanders (1), data are available for a total of 868 square kilometres, investigated during 2007-2009. Grey wagtails were only recorded in 7 squares. Grey wagtail is too sparse a breeding bird in Flanders and too much a localised habitat specialist to be monitored by the common breeding bird census, in which riparian habitat was furthermore not selected as a target habitat stratum.

The breeding population of Grey wagtail (2) used to be followed in the recording scheme of rare and uncommon breeding birds (BBV). However, when numbers increased to ca. 550 pairs the population had reached the threshold beyond which the species became too common to achieve an accurate follow up of breeding pairs annually. Consequently, it was recently dropped from the rare breeding bird scheme (Vermeersch, Anselin & Devos 2006).

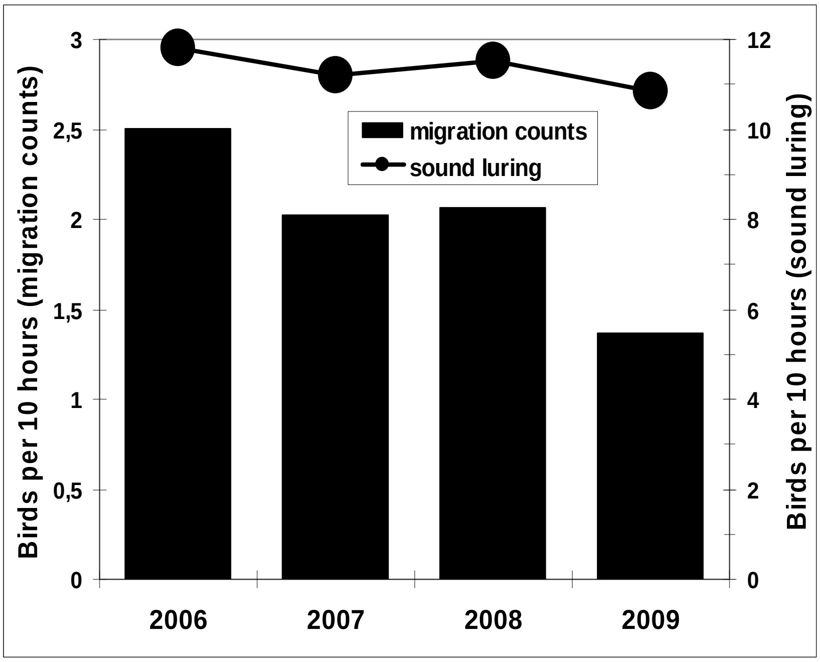

Figure 1. Results of Grey wagtail monitoring in Belgium during September-October: birds in migration counts (bars, left axis) and sound luring in a local ringing scheme (dots, right axis).

Over the last 20 years, 1856 mid-winter transect-counts (PTT) were carried out in Flanders (3). Grey wagtail numbers in these winter counts were very low: the species was only recorded in 61 counts, a total of 442 birds from 37 120 count-points. The average of ca. 9 records per winter is too low to evaluate annual changes. Grey wagtails prefer to winter in towns or elsewhere near buildings, where too few PTT-routes were established to monitor winter numbers effectively.Migration counts (4) are very popular in the low countries: credits for this go to Jethro Waanders and Gerard Troost, the developers of the website www.trektellen.nl, which has proved to be a fantastic and stimulating tool. Volunteers undertook counts of migrants at 80-94 locations throughout Flanders in recent years, recording 3.7-5.3 million migrants annually, mostly during September-November.

Grey wagtails are sparsely recorded during autumn migration counts, about two birds in 10 hours. In 2009, following the more severe winter, passage dropped by 33% compared to 2008. However, due to particular weather conditions in 2009 (prolonged periods with tailwinds), many migrants may have shifted to higher flight levels beyond the visible range as the overall number of migrants observed dropped by exactly the same 33%. Despite ca. 8000 hours of counts annually, it is therefore impossible to assess whether passage of Grey wagtails had actually declined in 2009.

In a local ringing scheme at an inland locality in Flanders (Scherpenheuvel)(5), I used continuous sound luring for five hours in the morning and two hours in the evening daily (weather permitting) during September and October 2006-2009. Grey wagtails invariably responded and eventually landed near a garden pond: all birds were counted, and 71% (815) were caught and ringed. Sound luring resulted in five times more birds per hour than migration counts, and numbers only dropped by 6% in 2009 compared to 2008 (Figure 1). However, among birds caught, the proportion of adults increased from 3.5% in 2008 to 9.4% in 2009, indicating that after the more severe winter a larger proportion of the population, including more adults, undertook migration the next autumn.

Most observers consider Grey wagtail an interesting species and records are readily reported to www.waarneming.be (6). Therefore, rates at which the species is reported do not deviate much from the rate at which it is encountered in the field.

The reporting rate of Grey wagtail dropped by 55% in Belgium between the month before the frost (15/11-15/12/2008) and the month after (15/1-14/2/2009), suggesting severe impact. In The Netherlands, a similar decline of 46% was recorded in www.waarnemingen.nl. Mortality could have been a major cause, but birds might as well still have escaped by emigration when the cold spell arrived.

Overall in 205.464 bird records during the breeding season (May-August), the reporting rate of Grey wagtails dropped in Flanders from 2008 to 2009 by 38% for the number of records and 40% for the number of birds, confirming severe impact on the local breeding population. For The Netherlands, the decline in reporting rates during the breeding season of 2009 was very similar: 36% fewer records and 40% fewer birds (in 892.637 records).

Discussion

In Flanders, naturalist observer density is among the highest in Europe. Despite substantial efforts in gathering various bird data in Flanders with well established and standardized recording scheme's, these do not suffice to assess winter impact on Grey wagtail numbers. Admittedly, most monitoring schemes were not designed to pick-up and account for annual (or even shorter-term) changes, but rather for long-term trends. Nevertheless, in winter-sensitive species like Grey wagtail the short-term dynamics induced by a severe cold spell can have large effects on the long-term trend and it is essential that they are somehow identified and quantified.

The ringing results showed that different fractions of the Grey wagtail population migrate in different years (more adults participating following a severe winter), which means that quantitative comparison of records outside the breeding season becomes very tricky in this partial migrant.

Specific breeding bird surveys are the best option to assess population trends in such partial migrants, particularly in habitat specialists. Unfortunately, we do not have a scheme in place that covers this in Flanders. It is a labour-intensive and time-consuming work anyway for such a sparse habitat specialist. In The Netherlands, some breeding populations of Grey wagtail dropped by 40% in 2009 (Boele 2009) and the overall population-index for the species based on breeding bird monitoring showed a drop by 35% (A. Boele, SOVON, pers. comm.).

Most interestingly, the drop in reporting rates amongst 'occasional records' during the breeding season in both Belgium and The Netherlands of 38-39% comes very close to the data from the breeding bird surveys in The Netherlands. Apparently, in this readily reported species, changes in reporting rates in large datasets of occasional records are a useful alternative to specific and standardized monitoring. These reporting rates are based on a dataset that furthermore requires no 'special effort'. It is worth investigating more in general for what species and conditions reporting rates amongst occasional records match monitoring data closely and could constitute an alternative, particularly for the documentation of short-term changes and the assessment of the events that caused them.

Acknowledgements

I am particularly grateful to the following persons for cooperation and communication of data under their curation: Arjan Boele (Sovon), Hisko De Vries (Stichting Natuurinformatie, www.waarnemingen.nl); Wouter Vanreusel (Natuurpunt, www.waarnemingen.be), Pieter Van Dorsselaer en Goedele Verbeylen (Natuurpunt, PTT) and Glenn Vermeersch (INBO, ABV).

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MS received: 10.08.2010
Revised MS accepted: 29.11.2011

Bird Census News 2010, 23/1-2: 117-123

The history of Belgian bird ringing and its potential for monitoring and migration research

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Abstract. This note summarizes Belgian ringing activities since the start of ringing in 1927. Today the scheme is supported by a large network of efficient and well-structured volunteer ringers, making Belgium the country in Europe with the second highest number of birds ringed per year (679,710 birds of 246 species and subspecies were ringed in 2008). The computerization of the ringing evolved over the last decade, making statistical analysis and mapping possible for the first time. All annual ring totals since 1960, all recoveries, and 95% of the recent detailed ring data are fully digitized. Old detailed ring data are digitized retrospectively. The past and present distribution of the ringing places, the geographical distribution of recoveries, and the changing numbers of birds ringed, illustrate the large potential of data collected for monitoring and topics related to bird migration.

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The development of the Belgian Ringing Scheme

The Belgian Ringing Scheme was launched in 1927 at the Royal Museum of Natural History in Brussels, today named the 'Royal Belgian Institute of Natural Sciences'. There were 76 volunteer ringers that collaborated at the beginning in September 1927, becoming 139 in 1928, and they ringed 4 425 birds during these two first years (Dupond 1928, 1929). The ringing total of the first decade was about 250 000, put on by about 460 volunteer ringers (Verheyen 1939). The annual total of ringed birds was 93 590 in 1960, reaching over 127 657 birds the year after (Arnhem 1963) and has grown continuously since.

The scheme was restructured in the early sixties. The ringers were requested to work in regional ringing groups or stations. Each ringing station had to match several conditions, one of which was to ring a minimum of 5 000 birds of 65 different species per year, for one station with 3 ringers (Arnhem 1964). No individual ringer was authorized from then. Every group or station was headed by a person in charge of the administration of the team and acting as the contact person between the ringers and the ringing office at the Institute. They had to compile the annual ringing total of the team and report to the provincial responsible in charge of the ringing coordination within his province. The provincial persons in charge was designated by the Institute among the local heads of ringing groups and stations. At the same time, the certification of all the ringers became mandatory. It was also among the duties of the heads of ringing groups and stations to organize the training of the candidate ringers (Arnhem 1961b, 1963).

These new rules and organization had a profound effect on the development of the ringing activity in Belgium. The structure in ringing groups and stations has allowed developing coherent trapping activity, ornithological investigations and administrative efficiency. It has created emulation between ringers for catching more various species and in larger numbers. At the beginning, most species ringed were finches and other favourite species of the traditional bird-catchers in Belgium.

The next major development of the Ringing Centre was the digitalization of data. The initiative for making the data available on disk and therefore increasing their accessibility for analyse was launched in 2000. PAPAGENO is the software developed in this framework to record ringing and recapture data, but also to make analyses including mapping. It was developed thanks to the essential contribution of a volunteer ringer, P. Vandenbulcke. Additional software to record and analyse the annual ringing totals was developed together with other basic analytical tools.

The network of Belgian ringers is nowadays 353 volunteer collaborators. No restrictions except those related to the welfare of the birds and the respect of administrative procedures are in force. All species may be ringed and all individuals caught have to be ringed. With more than 650 000 birds ringed per year, Belgium is second in Europe, behind the United Kingdom, amongst those with the highest number of birds ringed annually.

The successive heads of the Belgian Ringing Centre (Ch. Dupond, René K. Verheyen, Rudy Verheyen, W. Roggeman) have successfully maintained the motivation and personal investment of such a large network of ringers. They have published a part of the annual ringing totals and lists of recoveries in the Belgian ornithological journal ‘Le Gerfaut-De Giervalk’. Since the early eighties lack of personal has restricted the time devoted to the scientific analyse of the huge amount of data available. This is unfortunate given the present debates and studies on change in migrating routes of birds. This note aims to give recognition to the Belgian Ringing Centre, presenting the global state of the data collected.

The database

The data collected by the Belgian Ringing Centre are now stored in three electronic databases supplemented by the paper archives. The actual ringing database includes 8 378 732 records. For the last 5 years the proportion of computerized birds is over 96%. This remarkable result is due to good-willing of the volunteer ringers who encode their current but also their past data and in some cases even the data of others. Additional recording of some old ringing data has been made thanks to a grant given to the Institute by the Belgian Science Policy (program DIGIT 05). The rest of the ringing data estimated to 15 500 000 is available on paper archive only.

The recovery database includes 516 452 records (up to December 2009) and includes all the recoveries (for which ringer and finder are different) recorded between 1927 and today. Most of the recaptures where the ringer and finder are identical (own retraps) are included since 2000.

The third database contains the annual ringing totals since 1960. All of them are computerized, therefore we know that a grand total of 21 147 802 birds belonging to 409 species or subspecies have been ringed from 1960 to 2008 included.

This recent digitization of data allows for the first time analyses with modern computing tools.

Ringing activity

Mapping the ringing places according to the number of ringed birds for the whole time period with the data computerized so far shows the even spread of the ringing activity throughout the country (Figure 1). The geographical ringing locations refer to the smallest administrative units (the municipalities as before the 1977 fusion) rather than to the precise ringing locations. But some special well-known sites have also been considered within the municipality perimeter, and therefore there are some cases with two or rarely three dots inside one perimeter. Some parts of the country are less visited by ringers than others, especially the Ardennes area (Luxembourg province), but less than 15% of the municipalities have no single ringed bird in their territory among the data computerized so far for the whole time period since 1930 (oldest ringing data encoded to date). Most of the detailed data older than the eighties are still largely lacking in the electronic data base and form the invisible part of the 'ringing data iceberg'.

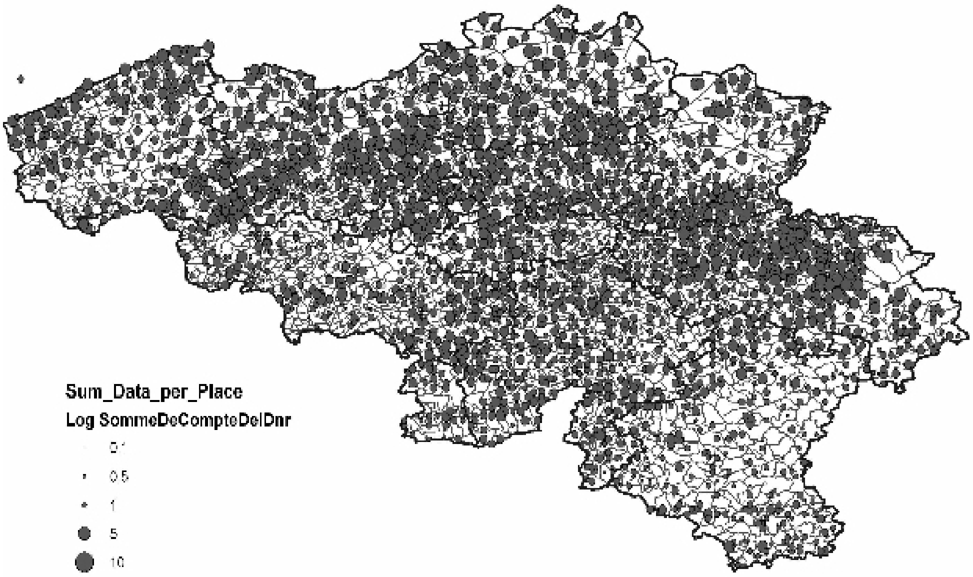

Figure 1. The number of ringed birds by site after the some of all data computerized to date shows that the country has been well covered

The ringing activity and trapping efficiency have grown since 1927. First by interesting the ringers to look for other species than the ones they initially trapped during the autumn migration for encaging (as it was allowed to encage), and then from the sixties, by using new developed catching techniques and means such as mist nets. Later on, the use of tape lure did gave a new impulse to the total sums of birds caught and ringed, and more again when they were used by night, from the mid eighties. The annual total graph of all ringed birds (Figure 2) reflects the development of both main trapping effort and global ringing activity.

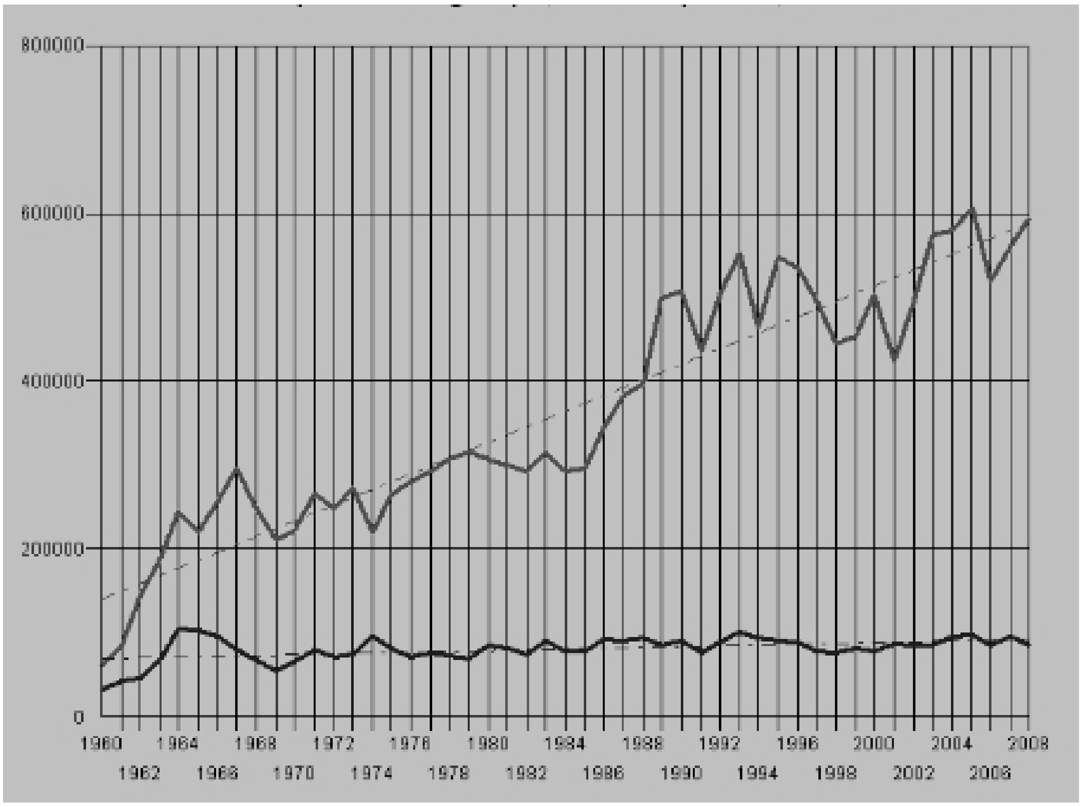

Figure 2. Evolution of the yearly ringed bird numbers from 1960 to 2008 (upper line full grown: total 17 976 599, lower line pulli: total 3 979 604, dotted line=regression line)

The ringing activity during the last decade

More than 350 ringers were active in Belgium the last decade. The trapping places are still widespread and provide a good sampling along the migration front crossing the country.

The most ringed species in Belgium

Table 1 shows the top 10 of the most ringed species in number as pullus, full-grown and the sum of both categories, for the whole time period since 1960, for the last decade and for the last completed activity year (2008). Seven species are in the global top 10 list over the three periods: Blackcap *Sylvia atricapilla*, Great Tit *Parus major* and Blue Tit *Parus caeruleus*, Reed Warbler *Acrocephalus scirpaceus* , Barn Swallow *Hirundo rustica*, Meadow Pipit *Anthus pratensis* and Robin *Erithacus rubecula*. Most of these species are trapped as full-grown during postnuptial migration.

The top 10 lists of birds ringed as pullus for the three periods have only 5 species in common for each period: the two tit species already cited above, Barn Swallow, Starling *Sturnus vugaris,* and Kestrel *Falco tinnunculus*. Species which nest in holes or human buildings are more often ringed as *pulli*. This is the case of Great and Blue Tits as well as some raptors like the Barn Owl *Tyto alba*, the Little Owl *Athene noctua* and the Kestrel. The installation of nest-boxes benefits them, and this habit is more widespread since the eighties. Changes among the five others species in the pullus list are either due to a declining interest to ring certain species and for others the growing use of nest boxes.

The top 10 species list for full-growns reflect the most ringed species, mainly during the autumn migration. The Skylark *Alauda arvensis* is represented in the three periods even if it is not in the global list for 2008. Linnet *Carduelis cannabina* and Chaffinch *Fringilla coelebs* disappear of the top 10 lists for the last decade and 2008 year. They are replaced by Chiffchaff *Phylloscopus collybita* and Redwing *Turdus iliacus*.

The various specific graphical patterns of the sum of ringed birds as pullus or full-grown by year since 1960 must be interpreted in relation to development of trapping effort and methods. But species for which no special method is required or done to catch them as adults in Belgium, such as Blackbird *Turdus merula* and Tits including the Willow and Marsh Tits *Parus palustris & montanus,* Coal Tit *Parus ater*, Crested Tit *Parus cristatus*, and Long-Tailed Tit *Aegithalos caudatus*, could show a yearly variation reflecting their fluctuations. This will not be discussed in this paper.

Recoveries

Belgium is situated within the main western migration route. There are more than 500 000 recoveries which show the input that Belgian ringing activity can bring to migration studies. Most recoveries are within Europe (including the USSR), but also the northern part of Africa, the sub-sahel zone and the southern part of this continent are important. The GIS tool allows detailed geographical comparisons of the recoveries. Extracting the recoveries by species, season and time scale provide a useful basis for a recent research project on change of wintering grounds of partial migrant species.

Conclusion

The Belgian Ringing Centre has developed substantially since the early sixties, which has produced a very large amount of data spread over eight decades and concerning almost all species occurring in the country. That the volunteer ringers are largely distributed on the national territory and that they are certified give much value on the quality of the data sets.

Until recently the ringing data were only available on paper. The large scale digitalization initiated in the early 2000's opens several perspectives concerning the long term evolution of bird populations and the effect of climate change on wintering site selection. This was recognized by the Belgian Science Policy that recently granted the Belgian Ringing Centre money to undertake ad hoc analysis.

Acknowledgements

We thank all the volunteer ringers who did and still do collaborate with passion in the collection of data and partly finance the system. We also express our thanks to Walter Roggeman, head of the Belgian Ringing Centre 1972-2007, for his permanent support. The Belgian Ringing Centre is a unit of the Royal Belgian Institute of Natural Sciences (Belgian Science Policy). This contribution is part of a research project on the change of short-migrant bird's wintering grounds based on the ringing recovery data (WE/36/J08), funded by the Belgian Science Policy appointing the first author.

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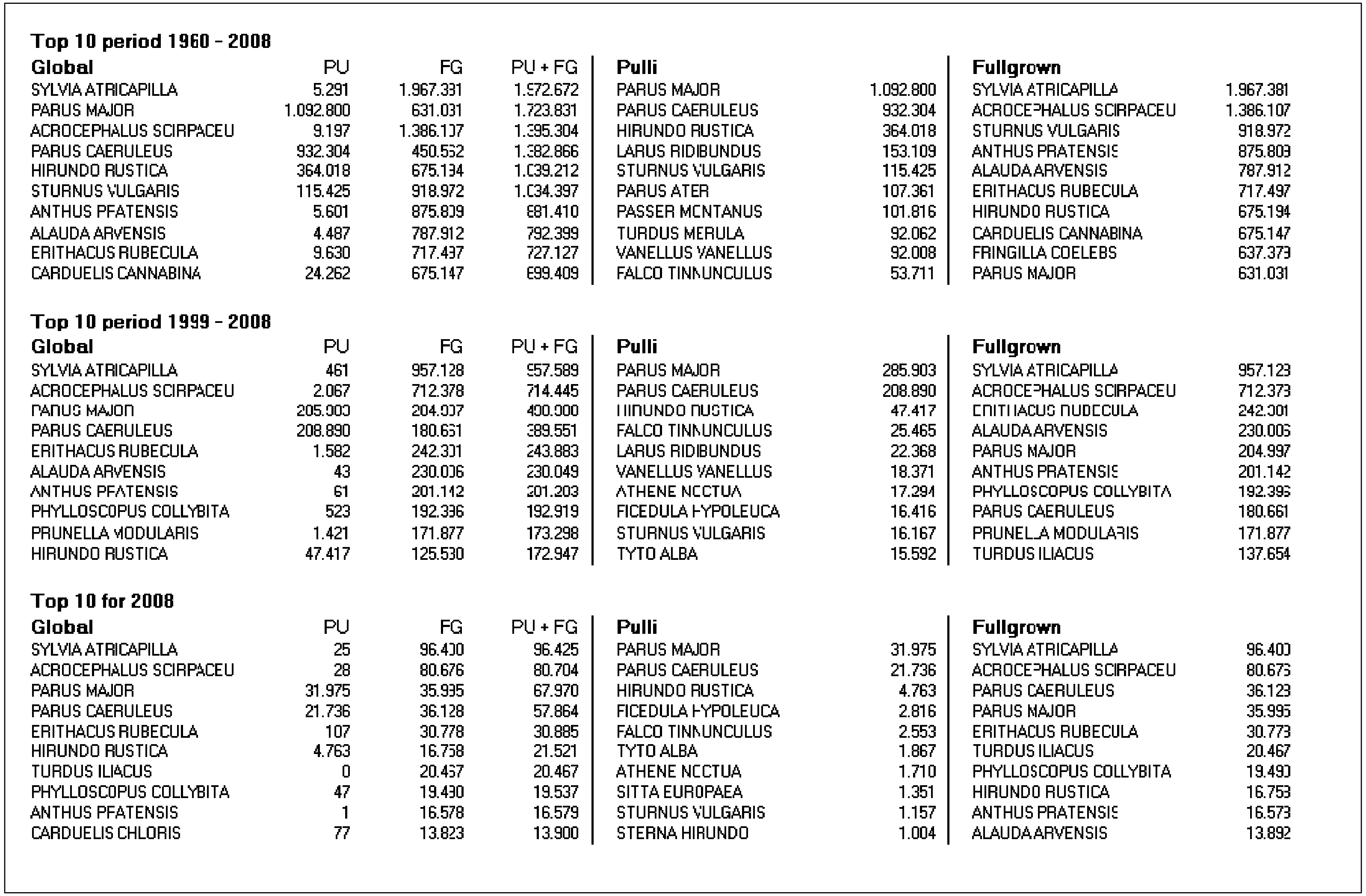
MS received: 11.05.2010
Revised MS accepted: 28.01.2011



Top 10 period 1960 - 2008

Global	PU	FG	PU + FG	**Pulli**		**Fullgrown**	
SYLVIA ATRICAPILLA	5.291	1.967.381	1.972.672	PARUS MAJOR	1.092.800	SYLVIA ATRICAPILLA	1.967.381
PARUS MAJOR	1.092.800	631.031	1.723.831	PARUS CAERULEUS	932.304	ACROCEPHALUS SCIRPACEU	1.386.107
ACROCEPHALUS SCIRPACEU	9.197	1.386.107	1.395.304	HIRUNDO RUSTICA	364.018	STURNUS VULGARIS	918.972
PARUS CAERULEUS	932.304	450.562	1.382.866	LARUS RIDIBUNDUS	153.109	ANTHUS PRATENSIS	875.809
HIRUNDO RUSTICA	364.018	675.194	1.039.212	STURNUS VULGARIS	115.425	ALAUDA ARVENSIS	787.912
STURNUS VULGARIS	115.425	918.972	1.034.397	PARUS ATER	107.361	ERITHACUS RUBECULA	717.497
ANTHUS PRATENSIS	5.601	875.809	881.410	PASSER MONTANUS	101.816	HIRUNDO RUSTICA	675.194
ALAUDA ARVENSIS	4.487	787.912	792.399	TURDUS MERULA	92.062	CARDUELIS CANNABINA	675.147
ERITHACUS RUBECULA	9.630	717.497	727.127	VANELLUS VANELLUS	92.008	FRINGILLA COELEBS	637.373
CARDUELIS CANNABINA	24.262	675.147	699.409	FALCO TINNUNCULUS	53.711	PARUS MAJOR	631.031

Top 10 period 1999 - 2008

Global	PU	FG	PU + FG	**Pulli**		**Fullgrown**	
SYLVIA ATRICAPILLA	461	957.128	957.589	PARUS MAJOR	285.903	SYLVIA ATRICAPILLA	957.128
ACROCEPHALUS SCIRPACEU	2.067	712.378	714.445	PARUS CAERULEUS	208.890	ACROCEPHALUS SCIRPACEU	712.378
PARUS MAJOR	285.903	204.997	490.900	HIRUNDO RUSTICA	47.417	ERITHACUS RUBECULA	242.301
PARUS CAERULEUS	208.890	180.661	389.551	FALCO TINNUNCULUS	25.465	ALAUDA ARVENSIS	230.006
ERITHACUS RUBECULA	1.582	242.301	243.883	LARUS RIDIBUNDUS	22.368	PARUS MAJOR	204.997
ALAUDA ARVENSIS	43	230.006	230.049	VANELLUS VANELLUS	18.371	ANTHUS PRATENSIS	201.142
ANTHUS PRATENSIS	61	201.142	201.203	ATHENE NOCTUA	17.294	PHYLLOSCOPUS COLLYBITA	192.396
PHYLLOSCOPUS COLLYBITA	523	192.396	192.919	FICEDULA HYPOLEUCA	16.416	PARUS CAERULEUS	180.661
PRUNELLA MODULARIS	1.421	171.877	173.298	STURNUS VULGARIS	16.167	PRUNELLA MODULARIS	171.877
HIRUNDO RUSTICA	47.417	125.530	172.947	TYTO ALBA	15.592	TURDUS ILIACUS	137.654

Top 10 for 2008

Global	PU	FG	PU + FG	**Pulli**		**Fullgrown**	
SYLVIA ATRICAPILLA	25	96.400	96.425	PARUS MAJOR	31.975	SYLVIA ATRICAPILLA	96.400
ACROCEPHALUS SCIRPACEU	28	80.676	80.704	PARUS CAERULEUS	21.736	ACROCEPHALUS SCIRPACEU	80.676
PARUS MAJOR	31.975	35.995	67.970	HIRUNDO RUSTICA	4.763	PARUS CAERULEUS	36.128
PARUS CAERULEUS	21.736	36.128	57.864	FICEDULA HYPOLEUCA	2.816	PARUS MAJOR	35.995
ERITHACUS RUBECULA	107	30.778	30.885	FALCO TINNUNCULUS	2.553	ERITHACUS RUBECULA	30.778
HIRUNDO RUSTICA	4.763	16.758	21.521	TYTO ALBA	1.867	TURDUS ILIACUS	20.467
TURDUS ILIACUS	0	20.467	20.467	ATHENE NOCTUA	1.710	PHYLLOSCOPUS COLLYBITA	19.490
PHYLLOSCOPUS COLLYBITA	47	19.490	19.537	SITTA EUROPAEA	1.351	HIRUNDO RUSTICA	16.758
ANTHUS PRATENSIS	1	16.578	16.579	STURNUS VULGARIS	1.157	ANTHUS PRATENSIS	16.578
CARDUELIS CHLORIS	77	13.823	13.900	STERNA HIRUNDO	1.004	ALAUDA ARVENSIS	13.892

Table 1. The top 10 of the most ringed species in number as pullus, full-grown and the sum of both categories, for the whole time period since 1960, for the last decade and for the last completed activity year (2008).

Bird Census News 2010, 23/1-2: 125-140

Breeding area, habitats and numbers of Common Snipe in European Russia

Yuri Yu. Blokhin

Abstract. From 2003 to 2009 a research project on the reproduction of the Common Snipe, *Gallinago gallinago* has been developed in Russia to improve the knowledge on distribution and population numbers. This first study shows that the future of the species in Russia will be highly dependent on the state of the mire habitats in the country. Weather conditions of the recent years affected the population, however no important declines have been noted hitherto. Population estimates in the study must be regarded as preliminary and more research in the future is needed.

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Introduction

The project “Research programme on the reproduction of the Common Snipe (*Gallinago gallinago*) in Russia” ran from 2003 to 2009, with the aim of developing research on this species, and in particular improving knowledge of its distribution area and estimation of the breeding population size. The research was carried out by the “Russian Bird Conservation Union” (RBCU), with coordination and financing from the French “Office National de la Chasse et de la Faune Sauvage” (ONCFS). In different seasons, between 12 and 23 participants took part in the project, totalling 44. The author of the present review was the Russian project coordinator.

Background of the study

Population estimates

The Common Snipe is a widespread bird species, but there are not enough data on the species’ numbers and biology in various parts of its range. These data are necessary for estimating the species’ abundance over larger areas. Hence, data collection on species’ numbers at various latitudes, based on application of unified methods, is very important.

In Europe, especially in France, many studies on Snipe have been conducted, and special attention has been given to estimating the size of different populations (e.g. Devort *et al.,* 1997; Rouxel, 2000). Determination of Snipe numbers is conducted during the breeding period, the phase in the annual cycle during which the birds are temporarily sedentary. Usually, the number of breeding pairs on a census ground is revealed by displaying males, and the precise number of nests on a plot is only rarely counted (Smith, 1981, Green, 1985). Thus the figures given in the national atlases of the various European countries are subject to imprecision, often, no doubt, to a considerable degree. Moreover, the numbers of Snipe breeding in many European countries are relatively small. Studies carried out in limited areas with low densities of birds are the only ones that can be as near to exhaustive as possible. No doubt this is the case for France, with figures totalling 200 to 300 pairs (Devort *et al.,* 1997).

In the Common Snipe Action Plan it was emphasised that the Russian, Polish and Byelorussian populations, which in large part supply the migratory flows in Western Europe, are not known (Devort *et al.,* 1997). Thus care should be taken when attempting to estimate the total size of the breeding population in the Western Palearctic. The task of collecting information on the number and distribution of Snipe in Russia is complicated by the huge size of the country. Extrapolating from local counts to estimate the species' national population size is clearly very challenging and carries a high risk of including a substantial error margin. Nevertheless, such attempts were already made in the end of the 20th century.

In the Atlas of European breeding birds numbers of Common Snipe in European Russia were estimated at around 1 - 10 million breeding pairs (Beintema & Saari, 1997), which made up 56 – 91 % of the total European population (Rogacheva & Syroechkovsky, 2003). The size of the breeding population in European Russia is not really known, except that it is perhaps between 300 000 and 4 000 000 pairs, as suggested by some sources (Rouxel, 2000). The most recent estimation for European Russia was made by E. Lebedeva and P. Tomkovich in the framework of BirdLife International's "Birds of Europe 2" project: they estimated it as 300 000 to 850 000 pairs, and suggested that the number of Common Snipe in Russia was probably stable during the 1990s (Mischenko *et al*., 2004).

Methodology

The inventory of the species was done using Breeding Bird Surveys (BBS) in May and June in 2002 and 2006. A total of 30 500 ha area of wetland and surrounding ecosystems were chosen for the study and was divided into 305 1x1km UTM (Universal Transverse Mercator) grids. In each grid, 3 random points separated by at least 300m distance were sampled and surveyed once. Point counts were conducted with a duration of 10 minutes each. The number of point counts in a square was reduced when the were largely covered by open water.

Apart from the standardised counts, all other casual observations of birds seen and/or heard and their breeding codes were noted separately. The number of individuals for each bird species observed at each point were counted and their breeding evidence was quantified using the international standardised breeding categories and codes (Hagemeijer & Blair 1997, Bibby et al. 2000). Threats and habitats were noted at each point. In 2006, 25 squares with a high coverage of open water habitat and/or urbanised area were omitted from the survey.

Habitat requirements

During the breeding period, the Common Snipe chooses all types of peaty marshes containing sedge (*Carex spp*.), rush (*Juncus spp*.), moss and *Sphagnum spp*. Areas with willows (*Salix spp*.), alders (*Alnus spp*.), birches (*Betula spp*.) and larches (*Larix spp*.) seem to be favoured (Rouxel, 2000). Due to the length of its beak (approximately 65 mm) and its legs (approximately 68 mm), the Common Snipe lives in areas where the waterdepth does not exceed 50 mm. It is the length of its beak, which enables it to probe the wet earth for food, and the length of its legs that determine the type of biotope that is ideal for this species (Devort *et al.* 1997). It prefers fairly open wetlands (where the birds can fly low), with short grass up to 25 cm, where Snipe can move in search of food and hide. At the same time, it is necessary for breeding that some elevated features (e.g. low hills) are present too. For feeding, Snipe should have direct access to soil, which must be soft, wet, peaty and rich in organic material (Rouxel, 2000).

As the majority of large bog areas in Western Europe have been drained and reclaimed, the Common Snipe now uses in this zone flooded fields changed by man, such as pastures and hayfields, where elevations alternate with depressions flooded with 1-3 cm of water. Contrary to mires, such habitats are highly sensitive to changes that affect their suitability for Snipe, because they can be drained or flooded easily (Devort *et al.,* 1997). Detailed research in Kaliningrad Region confirmed that the hydrological condition is the main factor affecting the numbers and distribution of breeding Snipes, although the height and density of grass are important as well (Yarovikova, 2004).

Breeding range

The breeding range of the Common Snipe is shown in Figure 1. The southern limit of the Snipe's distribution in European Russia coincides with the confluence of the rivers Severskiy Donetz and Don in Rostov province **(1)**. However, the species does not breed here every year (Petrov & Nachaev, 1987; Belik, 1999).

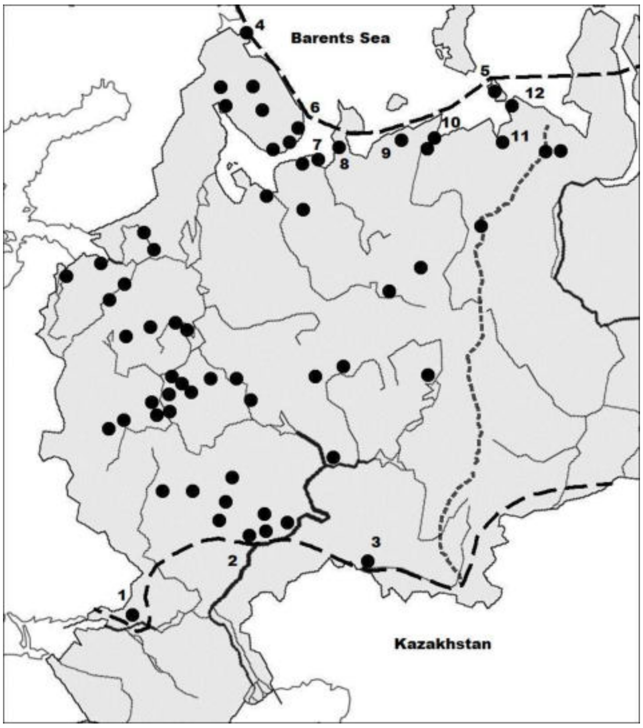

Figure 1. Locations with confirmed breeding of the Common Snipe and the border of breeding range in the European part of Russia (thick dashed line). The names of numbered localities are mentioned in the text below.

To the southeast, in the South Ural region, the Snipe is found only during migration in steppes along the rivers Kuma and Terek, in West and East Manych, in East Priazovye and Predkavkazye (Oleynikov *et al.*, 1973), and in the North Caucasus in Adygeya (Peklo, 1980).

The Lower Volga forms the south-eastern distribution limit in European Russia. The species breeds only in the north and north-west of the Saratov province **(2)**, the southern limit of the range lays 40 km upstream of Saratov. The most southerly breedings sites in the province are found at about 51° 40" N (Zavyalov *et al.*, 1998). More to the east, the species occurs in the South Ural in the Orenburg province **(3)**, although in the Svetlinsky district and the Sol-Iletsky district (51° N and 55° E) only passage has been observed (Davygora, 2000; Korshikov, 2000, Ryabitsev *et al.*, 2001).

In the Barents Sea the species breeds on the Aynovy islands **(4)** (Tatarinkova, 1980, 1983) and on Vaygach island **(5)** (Kalyakin, 1988). Since the 1990s there are observations including displaying birds on Novaya Zemlya (Kalyakin, 1998, 2001).

Kozlova (1962) mentions the presence of clutches and chicks in the Varanger fjord, close to the north-western frontier of Russia. Near the border with Norway on the Kola Peninsula (in the “Pasvik” reserve) breeding was suspected by Grachev (2002) and confirmed by Makarova *et al*. (2003). Nesting was confirmed in the narrow entrance of the White Sea **(6)**, in the lower river Ponoy (Bianki *et al*., 1982), at the Tersky shore of the Kola Peninsula (Malyshevsky, 1962, Goryaev, Krasnov, 2005) and on the coast of the Mezenskaya inlet **(7)** (Kozlova, 1962).

The breeding status of the Common Snipe north on Kanin Peninsula is still unknown (Yestafyev, 1995), but according to Vinogradov (1999) it breeds in the Torna-Shoyninskoye interfluvial zone **(8)**. It nests in Malozemelskaya tundra (Vel’t **(9)** and Indiga rivers) (Gladkov, 1951, Mineyev *et al*., 1999). It was found nesting in the delta of Pechora **(10)**, in the Bolshezemelskaya tundra in the river More-Yu **(11)** (Yestafyev, 1991) and on the shore of Yugorsky Shar strait **(12)** (Uspensky, 1965 *et al*.).

Thus, the northern border of Common Snipe distribution is situated between latitudes 70 - 71 on Aynovy and Vaigach Islands, while on the mainland Cape Bely Nos on Yugorsky Peninsula is the most northern locality. The southerly extreme, but probably isolated from the main breeding range, is located between latitudes 47 – 48 in the lower reaches of Seversky Donets and between latitudes 51 and 52 in the Lower Volga zone.

The species breeds in southern, typical tundra, and forest-tundra vegetation zones, all over the forest zone, in forest-steppe, along river valleys and in lake hollows in steppes. It is absent in arctic tundra, semi-desert or desert.

Methods

Study area

Between April and July during the years 2003 to 2009, breeding censuses of Common Snipe were conducted on the territory of 52 regions in 18 parts of European Russia (Table 1 & Figure 2)

	2003	2004	2005	2006	2007	2008	2009
Provinces	6	10	11	8	10	8	7
Regions	9	18	18	16	19	18	15
Census plots	35	122	105	156	209	183	110
Total area, ha	9071	10631	9268	6453	10372	8952	6868

Table 1: Census effort on Common Snipe in different regions of Russia in 2003-2009

Breeding habitats were visited in the basins of the largest rivers: Volga, Ob, Pechora, Severnaya Dvina, Zapadnaya Dvina and their tributaries. The number of plots varied annually from 35 to 209, and their total area from 65 to 106 km^2. Overall, the counts were made on 880 plots (including plots monitored for several years), with total area of 579km^2

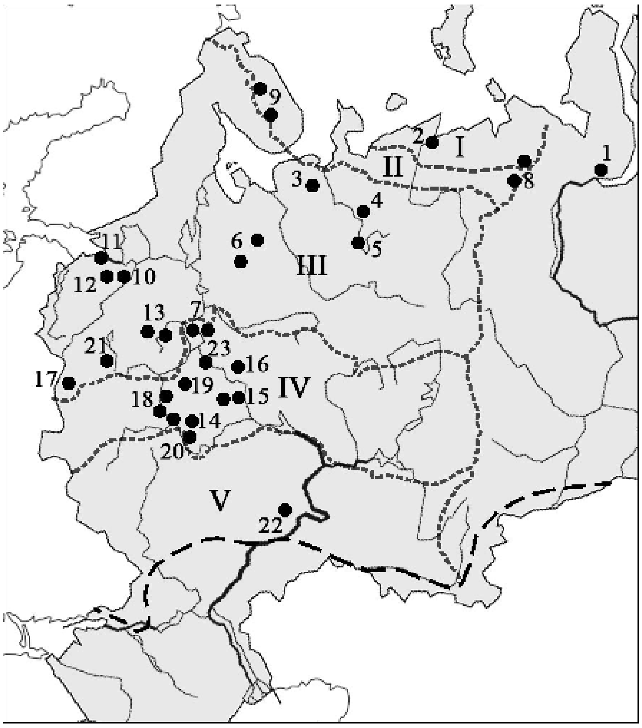

Figure 2: Study sites during the Snipe project in Russia. For names of localities we refer to Table 2

Information on the Common Snipe was collected from natural habitats ranging from southern tundra to deciduous forests (Table 2). Census plots were located more often in river valleys (in floodplains and areas between two rivers) and in interfluves. They included parts of swamps and meadows in combination with other, more often open, Snipe habitats. Human influence within the study areay was very heterogeous with as major activities cattle pasture, mowing and peat cutting. We collected general information on climate, hydrography and vegetation composition. In the specific census plots we determined the distinctive features of meso- and macro-relief, plant associations and the type of mire-vegetation. Weather conditions during the fieldwork were also noted.

Zone	**Sub-area**	**biotope**	**Bog type**	**unit**
Tundra	shrubby	FL, interfluvial, hilly	Open flat peat bogs	Komi (8), Nenetsky.(2), Yamalo-Nenetsky.(1)
Forest Tundra		FL, interfluvial	Open bog, low lying bog	Komi(8), Murmansk (9)
Forest	North Taiga	FL, interfluvial	Open and forest bog, humid meadow	Arkhangelsk(3), Komi (8)
	Middle Taiga	FL, interfluvial	Open and forest bog, humid meadow	Arkhangelsk(4,5,6)
	South Taiga	FL, interfluvial	Open and forest bog, humid meadow	Vologda(7), Leningrad (10,11), Ivanovo (15), Kostroma(16), Novgorod(12,13), Tver (7,19,21,23), Yaroslavl (7,23)
	Mixed	FL, interfluvial	Open and forest bog, humid meadow	Vladimir(14), Moscow Region (14,19), Ryazan'(20), Smolensk(17), Penza (22)
	Deciduous	FL	Open fen, humid meadow	Moscow Region (18)

Table 2. Vegetation zones, sub-areas, biotopes and bog types of the censused units. Numbers of units refer to Figure 2.

Census techniques

To estimate the number of nesting birds and census the population, a method based upon one of the European methods of Snipe counting was used (Smith, 1981; Green, 1985). This method was modified, taking into account the first results of research conducted in 2003 (Blokhin *et al*., 2004). It was tested in different geographical sub-areas. For this, information about optimal time of count during breeding period as a whole and during the 24-hour daily cycle was collected. Nests and broods were looked for using different means and methods. Plot sizes varied from 1 to 1200 hectares, with general size from 10 to 30 hectares. The majority of the plots were inspected several times during the breeding season.

The behaviour of territorial individuals was recorded. Repeated records of contacts with birds allowed the boundaries of individual display areas to be plotted on a large-scale map, revealing separate territorial males in places with high numbers of birds. It was conditionally accepted that one male, displaying over a permanent area, relates to one pair of Snipe.

Relative and absolute figures of breeding numbers were determined during censuses on grounds (conditionally breeding pairs/km^2). Average density of the population found in similar habitats was extrapolated to the total area of these habitats. The area of peat- and paludified lands were taken from the work of S.E. Vompersky *et al*. (2005).

Weather conditions 2003-2009

The 2003-2009 seasons differed between regions and resulted in variable breeding success. Weather anomalies were not infrequent. The weather conditions in the north were favourable in 2004, 2006 and 2008, but in the central region of Russia only in 2005. In the other years the conditions were unfavourable, affecting breeding success and numbers. Table 3 gives a summary of the phenology and weather conditions in the different regions during the seven years of the study period.

years	**North region**		**Central region**
	south tundra/forest-tundra	**north/middle taiga**	**south taiga/coniferous-deciduous forests**
2003	-	-	SP & eSU: cool and wet
2004	SU: dry and warm	SP:cold and wet	SP: cold
2005	eSP: early but cold spells	SP: dry and warm	SP: late, hot, floods
2006	SP: early, floods	SP: dry and cool	SP: late, dry and cool
2007	SP: cold; SU: dry and warm, high predator pressure	SP: cold, long	SP: early, dry; eSU:dry
2008	SP: late, SU: warm	SP: cold spells	SP: dry
2009		SP: cold and wet	SP: cold en dry

Table 3. General phenology and weather conditions in the three main regions of the study project (SU=summer, eSP=early spring, SP=spring).

Results

Breeding density in different habitats and regions

South tundra and forest-tundra (mire provinces with hilly bogs)

In Bolshezemelskaya tundra (r. Pechora basin), the range of breeding biotopes in the south tundra sub-area seems to be rather wide: from quite dry bush-moss tundra to open sedge fens with different transitions between them. Territorial birds, nests and broods were registered mainly on sedge-moss bogs, with brushwoods of thinned out bushes among flat-hilly peat-bogs or grass-sedge willow brushwoods along streams and river valleys, near lakes. Indices of Snipe population density differed greatly by biotopes and different breeding seasons (Table 4).

In forest-tundra (basins of r. Pechora and r. Ponoy on Kola Peninsula), Snipe displayed on hilly willow-sedge open fens in afforested river flood-lands and stream valleys, and on complex big-hilly peat-bogs with small lakes. On big-hilly bogs Snipe stayed on large flat plots with dominating bushes (*Cassandra calyculata, Vaccinium uliginosum*), undersized cotton grass (*Eriophorum medium*) and sphagnum mosses; however, in general, big-hilly bogs of forest-tundra seem to be weakly populated by Snipe.

In various tundra and forest-tundra landscapes, the most preferred Snipe nesting habitats are sparse and swampy grass-sedge osiers along river and stream valleys amidst complex flat-bumpy and large-bumpy peat bogs. Large-bumpy bogs are significantly less inhabited by snipes, because low-lying parcels are represented here mainly by sphagnous-cotton grass oligotrophic swamps, covering small areas and poor in food resources. At large-bumpy peat bogs, snipes settle mainly at patches with sparse shrubs near the sources of streams. Complex flat-bumpy bogs are predominating and cover large areas in southern tundra subzone, which is the most significant for breeding snipe in northern European Russia.

Types of habitats	Mean	On plots in different years
Flat-hilly peat bogs	12,2 ± 2,1	2,0 – 63,3
Big-hilly bogs	4,1 ± 0,8	0,9 – 10,0
Open fens (eutrophic)	15,0 ±3,5	10,0 –20,0
Flood-lands	11,8 ± 1,3	5,6 –20,0

Table 4. Breeding density of Common Snipe in swampy habitats (mire provinces 1-9) south tundra and forest-tundra, pairs/km^2

North, middle and south taiga (mire provinces of forests and convex bogs).

Snipe were numerous on big-hilly bogs with sparse bushes in stream valleys (basin of sredniaya Pechora). Displaying Snipe were found on maritime meadows (Gulf of Finland), on damp hilly flood-land meadows, on active or deserted hayfields, and on pastures. Highest densities were registered on flood-land meadows combined with open fens (on a number of plots in the basin of r. Severnaya Dvina), as well as on swampy forest clearings and overgrown burned areas. However, it was widely distributed in mesotrophic mire (forest and open) and mesotrophic parts of extensive raised bogs, which turned out to be the most important Snipe habitats. The species was frequent on open fens and in waterlogged forests: fir, osier and alder (Table 5).

Types of habitats	Mean	On plots in different years
Big-hilly bogs	16,1±6,6	5 – 33,3
Raised bogs (oligotrophic)	3,9 ± 0,8	0,3 - 20
Mesotrophic mire	4,3 ± 0,7	1 – 40
Open fens (eutrophic)	3,5 ±0,4	1,1 – 20
Forest transition bogs and waterlogged forests	4,3 ± 1,1	1,8 – 18,2
Water meadows and wetlands	3,5 ± 0,3	0,2 – 83.3
Forest swamped glades	5,1 ± 0,7	0,5 – 38,5
Swampy depressions in agricultural land/ along roadsides	5,1 ± 0,7	2,7 - 10

Table 5. Breeding density of Common Snipe in swampy habitats (mire provinces 11- 24) north, middle and south taiga, pairs/km^2

South taiga and coniferous-deciduous forests (mire provinces of eutrophic and oligotrophic pine-sphagnum bogs of Eastern Europe

The species seems prefer riversides, lakesides, shores of storage ponds, pools, former riverbeds and temporary reservoirs. On individual plots in flood-land habitats, where water meadows (hayfields, pastures, unused meadows) alternate with open sedge fens, a very high density of Snipe population was registered. The highest density of displaying males was on unused pastures near temporary reservoirs (in the valley of r. Oka). Density was high on a number of forest lakes and in temporarily flooded areas. In meadow and bushy areas of flood-lands, Snipe is attracted by forest edges. It is usual on watersheds, in some areas numerous, on open fens and forest fens and on vast burned areas, on depleted moderately wet peatbogs. The lowest density of breeding Snipe was on raised bogs (Table 6).

Types of habitats	**Mean**	**On plots in different years**
Raised bogs (oligotrophic)	1,2 ± 0,5	1,3 – 6,7
Mesotrophic mire	5,8 ± 1,4	1 – 11,9
Open fens (eutrophic)	10,9 ± 5,5	2,9 – 33,3
Forest fens and waterlogged forest	6,6 ± 1,1	0,8 – 33,3
Water meadows and wetlands	7,9 ± 0,6	0,7 – 60
Burnt-out places	11,9 ± 1,9	3,7 – 17,9
Swampy depressions in agricultural lands/ along roadsides	5,2 ± 0,9	1,4 - 8
Peat-hag	7,5 ± 0,8	2,7 – 16,4

Table 6. Breeding density of Common Snipe in swampy habitats (mire provinces 33-35) south taiga and coniferous-deciduous forests, pairs/km^2

Numbers of Common Snipe

To extrapolate Snipe census data, information was used from Vompersky *et al.* (2005), on distribution and area of peat-bogs of different categories (within the breeding area of the species) (Table 7). The original numeration of European mire provinces by Kats (1971) was preserved, but we do not cite the names of these provinces. The borders of mire provinces mainly match the borders of natural zones (tundra, forest-tundra, taiga), but do not match the borders of sub-areas.

Uneven distribution of resources utilized by Snipe on large territories could be explained by uneven distribution of its major habitats on a macroscale. Those most widely distributed on the Russian Plain are waterlogged forests, as well as oligotrophic (raised bogs), eutrophic (fens) and hilly bogs. However, the major resources used by Snipe are located on eutrophic and hilly bogs. In comparison with bogs, waterlogged meadows and flood-lands (with meadow and tree-bush vegetation) occupy a small area. Hilly bogs are distributed mainly in forest-tundra; high moors and mesotrophic mire forest or tree-less bogs dominate in taiga zone. In south taiga and coniferous-deciduous forests, open fens (grass, sedge, reed, etc.) are especially well developed as well as forest fens (alder, fir, etc.). Areas with peat thickness up to 30 cm are rated as paludified lands: tundra and forest-tundra, forests and sparse growth of trees and also flood-lands and meadows.

The number of Snipe on different categories of peatlands, regarding their distribution by groups of mire provinces, is presented in Table 8. On the basis of this census, we estimate the total peatland population at 1 080 000 to 1 840 000 breeding pairs. In paludified lands, we estimate Snipe numbers only in flood-lands and on meadows, where it breeds in numbers from 62 000 to 73 000 pairs. Thus, we estimate the total Common Snipe population in European Russia at between 1 145 000 and 1 913 000 pairs - 54,5% in tundra and forest-tundra zones, and 45.5% in bogs of forest zone.

Categories (groups)	**Mire provinces**					
	1 - 3	**5 - 9**	**11 - 24**	**33 - 35**	**39 - 40**	**Total**
			Peatlands			
Total	**45.6**	**44.8**	**104.3**	**16.7**	**1.5**	**212.9**
Polygonal bogs	1.5	0.1				1.6
Hilly bogs	16.5	26.1	13.2			55.8
Oligotrophic **	0.8	11.0	75.7	6.8	0.2	94.5
Mesotrophic	0.9	0.7	9.9	1.5		13.0
Eutrophic	25.9	6.9	5.5	8.5	1.3	48.1
			Paludified Lands			
Total	**39.2**	**33.8**	**278.1**	**20.6**	**0.7**	**372.4**
Tundra and forest-tundra	37.6	13.1	>0.1			50.7
Forests and sparse growth of trees		20.5	270.5	18.2	0.6	309.8

Table 7. Total area of peat- and paludified lands by mire provinces in European Russia,* thousands km^2

Categories (groups)	**Mire provinces**					
	1 - 3	**5 - 9**	**11 - 24**	**33 - 35**	**39 - 40**	**Total**
			Peatlands			
Total	**465-715**	**173-275**	**386-686**	**58-162**	**1**	**1083-1839**
Hilly bogs	167-236	86-128	125-300			378-664
Oligotrophic		8-19	208-315	5-12		221-346
Mesotrophic			36-50	7-11		43-61
Eutrophic	298-479	79-128	17-21	46-139	1	441-768
			Paludified Lands			
TotalL	**482-736**	**175-278**	**410-715**	**76-182**	**2**	**1145-1912**
Flood-lands and meadows	17-21	2-3	24-29	18-20	1	62-73

Table 8. Limits of breeding numbers of Common Snipe in peat- and paludified lands by mire provinces in the European part of Russia, thousands of breeding pairs

Discussion

During the breeding period, the population of Snipe is distributed diffusely and unevenly over a large part of European Russia, according to the local distribution of micro-habitats preferred by the species. The number of territorial males recorded during the census and the determination of density per unit of area are directly connected with the species' display activity. On the basis of daily observations of Snipe display activity in different latitudes, the optimum time was established with reference to local conditions. For example, in south tundra by Vorkuta in (67°N), the optimum census times under conditions of the "polar day" are from 04:00 to 09:00 in the morning and from 18:00 to 21:00 in the evening, according to local solar time. In the forest zone by Moscow (56°N), the optimum census times are from 04:00 to 05:00 and from 06:00 to 09:00 in the morning (showing an activity gap during the coldest period being just before sunrise), and also from 22:00 to 00:00 in the evening according to local time.

Regular censuses at the same sites allowed us to determine the number of territorial males (conventionally – breeding pairs) with some degree of accuracy. In separate periods, the census results demonstrate relative stability of the number of territorial males. However, this level often shows noticeable fluctuations during the species' long reproductive season (Figure 3)

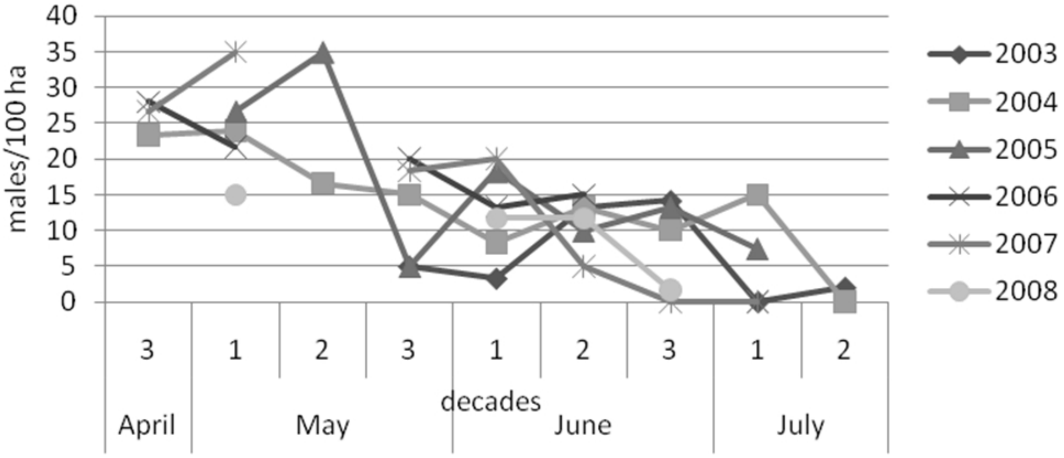

Figure 3. Seasonal and annual changes of Common Snipe density index on a 60 ha plot in the floodlands of r. Dubna

This phenomenon relates to variation in the activity of individual males' display, reflecting differences in the timing and success of breeding by different pairs. For example, in the Moscow Region we registered full new clutches as far apart as 10 May and 13 June 2004 (and according to the literature, this difference may be two months). Thus, while some males keep displaying actively, others are completely occupied with raising chicks and becomes less territorial, i.e. do not display. They are only rarely registered during censuses. Throughout the breeding range, the number of territorial birds shows a decreasing trend towards the end of the breeding season.

Within the same site, the number of Snipe show annual variations (Figure 4). In order to judge the scale of these changes correctly, we need information from many parts of the Snipe's natural habitat in different regions. Its importance is demonstrated in Figs 4 and 5, where the trends vary between regions. Censuses in areas where only 1 or 2 territorial males are registered annually also give a clear idea of annual variations of the number and distribution of Snipe in different habitats (presence - absence).

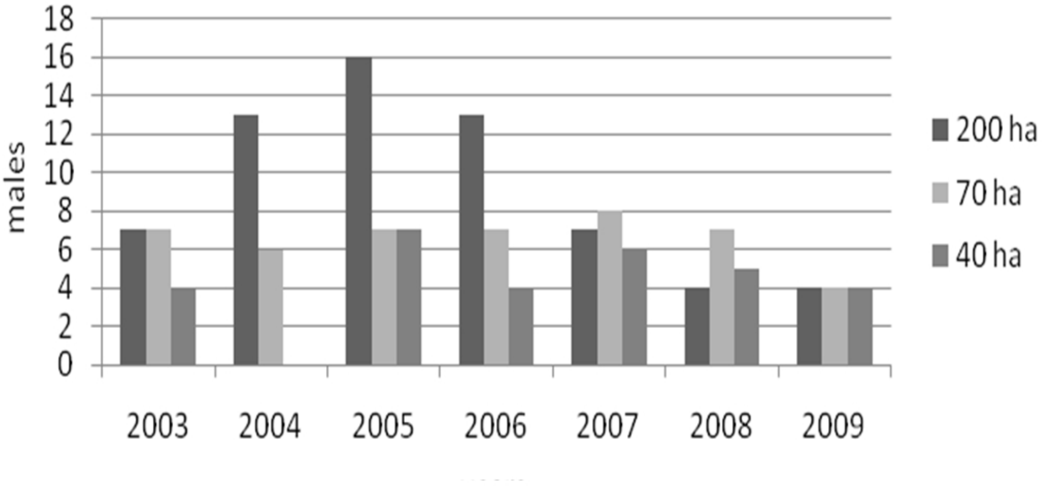

Figure 4. Numbers of territorial Common Snipe males on 3 permanent plots in Vladimir province (2003-2006)

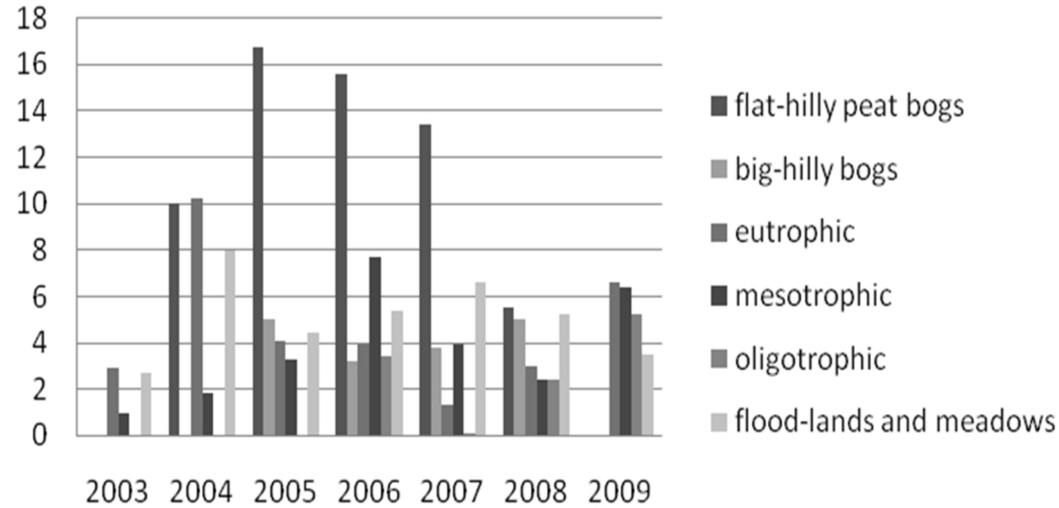

Figure 5: Dynamics of breeding density of Common Snipe in floodlands and mires of the Russian Plain in 2003-2009, pairs/km^2

According to the methods, it was recommended to conduct censuses only under favourable weather conditions, in order to obtain more objective figures. The effect of weather on display activity of Snipe and census results is shown in Figure 6, where weather characteristics, grouped by a number of main features (precipitation, wind, etc.), are evaluated as “good”, “satisfactory” and “bad”. At the same time, the effect of single weather factors is not always well reflected in the census results, and is analyzed in more depth in a separate paper (Blokhin *et al*., 2009).

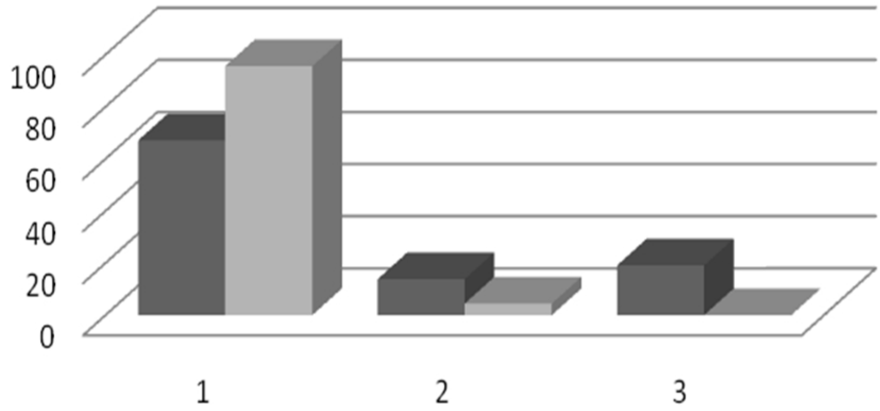

Figure 6. Dependence of Common Snipe activity on weather conditions (1=good, 2=satisfactory, 3=bad) in South Yamal

The major problem of obtaining good estimations of Snipe numbers even in small areas is caused by the difficulty of finding the nest which are well hidden. As a result, we have to estimate the number of breeding birds not by the number of nests found, but by the number of displaying males, whose activity varies strongly in space and time. Moreover, the density of Snipe in apparently similar habitats may be very different, as confirmed by information from censuses and the literature**.**

Comparing the size of the area censused and the area of the natural breeding habitat of the Common Snipe within Russia convinces us of the need for further work and data collection, including in other, still unexplored, regions. The absence of precise information for most of the country on the areas of fens, mesotrophic mires, wet meadows, swampy clearings, burned areas and certain other snipe habitats is a serious problem for estimating its population size and establishing trends. Quite often these habitats are sparsely distributed and cover relatively small areas, but in combination they are significant. Without more information on them, extrapolating census data on territorial males over the whole area of all categories of peatlands inevitably carries a risk and means that there is substantial margin for error when estimating the species' total population.

The distribution of Snipe is connected mainly with peatlands and marshland, which include a wide range of habitats for this species. The most important indications that bogs and other swamped habitats are suitable are their hydro- and source conditions. On oligotrophic, mostly sphagnum bogs (forest and open), Snipe are most often attracted by damp margins with mesophyte vegetation, and on hilly bogs to watered moss-sedge places.

Areas of different importance as Snipe breeding habitats are considered to be marshlands in the reference cited above. Only flood-land and meadow areas are presented individually by provinces. Drained forests, peatbogs, low forest in cleared areas, etc. were rated as waterlogged forests or forests with sparse growth of trees. This made it impossible to use the obtained indices of Snipe population density in these habitats for population estimation. Moreover, there is no information on the area of marshlands without peat bogs or excessive wet mineral soils, although, they are known to be habitats of the species.

For these reasons, and despite the advances made by this project, there is a lack of information from various bog habitats. It is also necessary to conduct censuses on polygonal mires and on ridge-depression complexes, to clarify Snipe numbers in paludified lands (except flood-land). At present, therefore, the estimated total population of Common Snipe in this study must be regarded as the best yet, but still preliminary and incomplete.

Conclusion

Peatlands and marshland are one of the most widespread landscapes in Russia, making up a substantial part of the Eurasian mires (Vompersky *et al.*, 2005). As far as peatlands are known to be the major breeding habitat of Common Snipe, the future of the species is highly dependent on the state of mires in Russia. In the review presented here, an attempt was made to estimate Snipe numbers in bog habitats, using areas of peat bogs and peat lands in European Russia. It will become more accurate as additional information is collected. Our estimations are not only based on expert opinion, but on census results from many regions and various natural conditions and in different breeding seasons. Annual monitoring shows that Snipe numbers fluctuate substantialy. The last years were abnormal in terms of weather conditions during the breeding season, which affected the state of the Snipe population in different parts of Russia in different ways. However, we did not note a decline of numbers.

Acknowledgements

This work became possible thanks to successful cooperation of ONCFS (coordinator Dr. Y. Ferrand) and RBCU, and the participation of a big team of ornithologists. The author thanks all participants of the project, especially: Dr. S. Yu. Fokin, A. Yu. Blokhin, Dr. V.G. Borschevsky, O. S. Grinchenko, M. V. Kozlova, Dr. V. N. Mel'nikov, A. P. Mezhnev, Dr. A. L. Mischenko, V. V. Morozov, Yu. M. Romanov, Dr. M.V. Sidenko, Dr. O. V. Sukhanova, Dr. T.V. Sviridova.

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MS received: 20.07.2010
Revised MS accepted: 04.02.2011

Bird Census News 2010, 23/1-2: 141-143

The Danish IBA Caretaker Project's assessment of the conservation status of Danish IBAs 1980-2010

Anne Eskildsen, Mathias Vogdrup-Schmidt, Timme Nyegaard & Thomas Vikstrøm

Abstract. Despite using two different methods of assessment, a similar number of Danish IBAs are assessed as having a favourable status (i.e. a score of 3) and an unfavourable status (i.e. a score of 1). However, there are extremely large differences in the number of IBAs assessed as having a quite favourable status (i.e. score of 2) and a very poor status (i.e. a score of 0). The main reason for an unfavourable conservation status of many IBAs is a decline of trigger species breeding in open habitats such as meadows, heaths and coasts. These habitat types are often threatened by overgrowing and disturbance.

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DOF caretakers monitoring the 128 Danish IBAs

The figures below summarize the development of the *trigger species* of the Danish IBAs since DOF's site surveys around 1980 and until today. The list of trigger species includes all species of breeding and staging birds, which according to BirdLife International's and/or EU criteria qualify a site as being of international importance.

Assessing the conservation status of the Danish IBAs

Figures 1 and 2 show the present conservation status of each IBA compared with its status around 1980. Comparison is made with this particular period because it was during this time that the EU Birds Directive came into force. It is therefore also this period which is used by the Danish nature authorities when assessing whether important bird areas are fulfilling the EU's demands for a *favourable conservation status*.

In each IBA, the status score of each species is calculated by using BirdLife's 0-3 point score system (BirdLife International 2006):

- A score of 3 indicates that the present population size is >90% of the population size in 1980
- A score of 2 indicates that the present population size is 70-90% of the population size in 1980
- A score of 1 indicates that the present population size is 40-70% of the population size in 1980
- A score of 0 indicates that the present population size is <40% of the population size in 1980

DOF has assessed conservation status in two ways

DOF prefers to assess the status score of an IBA by calculating an overall average using *all* trigger species. For each species in each period the status is measured by using the *highest number* of breeding or staging birds registered.

For comparison, DOF has also applied the *'weakest link'* principle recommended in BirdLife's global framework (BirdLife International 2006). Here the status score of the *individual* trigger species that is doing *most poorly* at the site is used to set the overall score for the site.

In Figures 1A and 1B the results of using these two different approaches are compared.

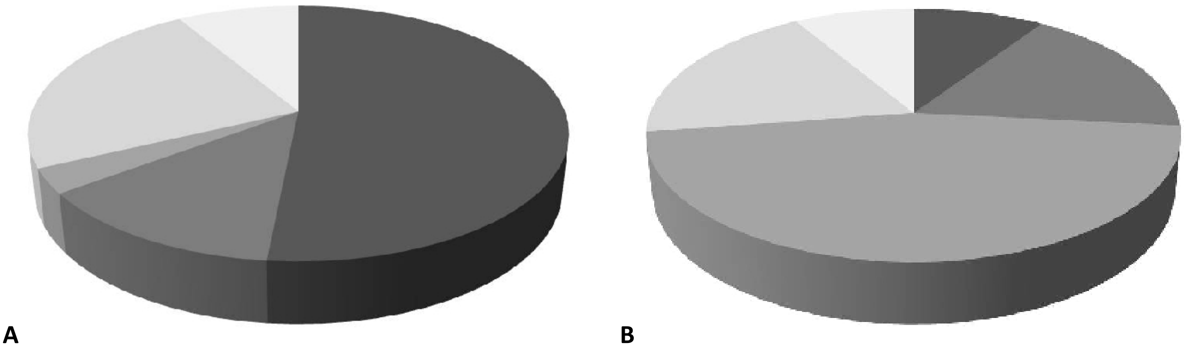

Figure 1. IBA status using the 'weakest link' approach (A) and using the 'average score' approach (B). For categories see legend below

■ Present population size is < 40 % of the population size in 1980

■ Present population size is 40-70 % of the population size in 1980

■ Present population size is 70-90 % of the population size in 1980

■ Present population size is > 90 % of the population size in 1980

□ Unknown

20 % of Danish IBAs have favourable conservation status

Figures 1A and 1B show that, despite using two different methods of assessment, a similar number of IBAs are assessed as having a favourable status (i.e. a score of 3) and an unfavourable status (i.e. a score of 1). However, there are extremely large differences in the number of IBAs assessed as having a quite favourable status (i.e. score of 2) and a very poor status (i.e. a score of 0).

Breeding birds of open habitats are decreasing while birds of other habitats are increasing

The main reason for an unfavourable conservation status of many IBAs is a decline of trigger species breeding in open habitats such as meadows, heaths and coasts. These habitat types are often threatened by overgrowing and disturbance. Population declines are most severe for waders such as Southern Dunlin *(Calidris alpina* ssp. *schinzii)*, Ruff *(Philomachus pugnax)*, Black-tailed Godwit *(Limosa limosa* ssp. *limosa)*, Wood Sandpiper *(Tringa glareola)*, Golden Plover (*Pluvialis apricaria)* and also for most tern species *(Sterna/Chlidonias* spp.*).*

Trigger species contributing to a favourable IBA conservation status are mostly birds breeding in forests and bogs such as Bittern *(Botaurus stellaris),* Greylag Goose *(Anser anser),* White-tailed Eagle *(Haliaeetus albicilla),* Red Kite *(Milvus milvus),* Marsh Harrier *(Circus aeruginosus)* and Common Crane *(Grus grus)*.

Staging diving ducks are decreasing, while geese are increasing

Staging bird decreases are found particularly among diving ducks such as Tufted Duck *(Aythya fuligula),* Greater Scaup *(A. marila)* and Goosander *(Mergus merganser)*. This may be due to water pollution and/or changed migration patterns.

Furthermore, some of these species are also contributing to a favourable IBA conservation status as staging birds, especially geese, of which Barnacle Goose *(Branta leucopsis)* has shown a remarkable increase.

References and databases

A large part of the data underlying the figures has been gathered by participants of the Danish IBA Caretaker Project. The majority of these data have been entered into the DOF database and BirdLife's World Bird Database.

A comprehensive reference list and more information can be acquired by contacting the DOF IBA Caretaker Project via thomas.vikstroem@dof.dk or tel. no. +45 3328 3822. The final report, detailing the results for each site, will be published in 2013.

Acknowledgements

The Danish IBA Caretaker Project is supported by the Aage V. Jensen Charitable Foundation.

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MS received: 12.05.2010
Received MS accepted: 20.01.2010

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