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# Using birds as indicators of biodiversity

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Here we develop a new method for reducing information on population trends for a large number of bird species into a simple composite indicator. This approach standardises species trends and creates a mean index across species. The 'headline indicator', which incorporates information on all common native species in the UK, is then disaggregated by habitat to reveal the underlying trends. The purpose of this paper is to introduce the method, discuss the most desirable qualities of wildlife indicators, to illustrate the practical difficulties in bringing such information together, and to show how the method can be used and developed.

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## 1. Introduction

Composite indices or indicators have many desirable properties, foremost being the reduction of complex information into simple visual summaries. In the field of economics the stock market indices, such as the Dow Jones, FTSE 100 and Nikkei, are highly familiar. Experts and non-experts alike understand the trends in these indices, though they may be unaware of the complex patterns shown by the underlying data.

There are however, few examples of such high profile and widely accepted indicators for biodiversity, although there are encouraging signs of change. Catalysed by the Rio de Janeiro Earth Summit in 1992, which reinforced the importance of biodiversity monitoring, a range of organisations have been involved in the development of indicators. For example, the Secretariat of the Convention on Biodiversity, United Nations Environment Programme (UNEP), the UN Commission on Sustainability (CSD), the World Bank, the Organisation for Economic Co-operation and Development (OECD), European Environment Agency (EEA) and BirdLife International. A series of recent studies have sought to clarify the role of environmental indicators and generate new indices (Kuik & Verbruggen 1991, Ten Brink *et al*. 1991, Reid *et al*. 1993, van Strien 1997, 1999; Ten Brink 1997, Bell & Morse 1999).

First, it is important to distinguish between 'state', 'driving force' and 'response' indicators. The first describes the state of a variable, the driving force gauges a process that influences the state, and the response measures specific actions to return the state to a desired condition. In this paper, we restrict discussion to quantifying a state indicator, namely, the

Tab. 1. Some of the desirable features of a wildlife indicator.

| Feature | Details |
|---|---|
| Representative | Includes all species in a chosen taxon, or a representative group |
| Immediacy | Capable of regular updating, *eg* on an annual basis |
| Transparency | Simple and easy to interpret |
| Assessment | Shows trends over time |
| Sensitivity | Sensitive to environmental change |
| Timeliness | Allows the timely identification of trends |
| Precision | Uses the raw data rather than categorical grouping of data |
| Cost | Does not require excessive financial resources to be produced |
| Available | Quantitative data are available |
| Indicative | Indicative of the more general situation among other taxa |
| Relevant | Policy and ecosystem relevant, relating to key sites and species; reflect main causes of biological change and conservation actions |
| Stability | Buffered from irregular, large natural fluctuations |
| Tractable | Susceptible to human influence and change |

population trends of breeding birds in the United Kingdom (UK).

There are a number of key attributes to effective bio-indicators. They must be; quantitative, simplifying, user driven, policy relevant, scientifically credible, responsive to changes, easily understood, realistic to collect, and susceptible to analysis (see Tab. 1).

One basic approach to generating an indicator of the state of wildlife is to measure diversity through time. Species loss or gain could then be used to gauge the trends in biodiversity. A problem with this method is that abundance and range could be modified without a net change in species number (van Strien 1997). There is also the problem that species of conservation concern may be supplanted by less desirable species, but in the process no overall change occurs in species diversity.

A second approach would be to determine the passage of species through categories of conservation status, *e.g.* IUCN categories (IUCN 1996). Van Strien (1997, 1999, see Discussion) has developed a refined version of this approach. If one's interest is in rare or endangered species then this method may be appropriate, but there are limitations. For example, it does not relate to biodiversity targets and can only be updated at fixed time intervals. Clearly, it does not take account of the status of common and widespread species in the environment.

A third approach would be to use a mean index of change taken across species. This would fulfil several of criteria for a wildlife indicator (Tab. 1). However, by taking this inclusive approach there is the potential for the declines among threatened species to be balanced by population gains among commoner, 'less desirable' species (van Strien 1997). The advantage is that it is transparent, and beyond scaling the population trends, no further decisions need to be made about choosing species, deciding on conservation status, nor deciding on population targets or reference periods. Therefore, while this approach has its shortcomings, there is much to admire in its simplicity.

Here we use birds as exemplar taxa to illustrate some of the issues in developing meaningful indicators for wildlife. We describe a new method for creating indicators based on the mean index.

Tab. 2. Sources of data for the indicators.

| Data source | Partners | No of spp | Units used |
|---|---|---|---|
| ATLAS | BTO/SOC/IWC | 42 | 10-km squares in UK |
| CBC | BTO/JNCC | 69 | CBC index 1970-97 |
| WBS | BTO/JNCC | 4 | WBS index 1974-96 |
| RBBP | BB/JNCC/RSPB/BTO | 51 | Mostly max total pairs 1973-95 |
| RBBP/SURVEY | RBBP/RSPB/EN | 4 | Mostly max total pairs 1973-95 |
| SCR | JNCC/SEABIRD GROUP | 9 | Pairs |
| SCR/SURVEY | SCR/RSPB/SNH | 3 | AOTs/pairs |
| SCR/SMP | JNCC/RSPB/SOTEAG | 4 | Mostly Thompson index |
| SURVEY | RSPB/JNCC/BTO/+ | 14 | Various |
| OTHER | BTO/RSPB/SNH/WWT/+ | 9 | Various |
| OTHER/SMP | | 2 | Various |
| OTHER/SURVEY | | 1 | Breeding pairs |
| WEBS | WWT/BTO/JNCC/RSPB | 6 | WeBS index, 1970/71-1996/97 |
| GAME BAG | GCT | 1 | Bag / 100ha |
| NONE | | 10 | |

*Acronyms of data sources explained in text.*
BTO = British Trust for Ornithology. SOC = Scottish Ornithologists' Club. IWC = Irish Wildbird Conservancy (now BirdWatch Ireland). JNCC = Joint Nature Conservation Committee. BB = *British Birds*. RSPB = Royal Society for the Protection of Birds. EN = English Nature. SNH = Scottish Natural Heritage. SOTEAG = Shetland Oil Terminal Environmental Advisory Group. WWT = Wildfowl and Wetlands Trust. GCT = Game Conservancy Trust. + = various other sources. AOT = apparently occupied territories.

## 2. Methods

### 2.1. Data sources

Taking an inclusive approach to producing a wild bird indicator, we first interrogated all of the long-term bird data sets to obtain information on population trends or range changes for as many species as possible. Because the dates of the first breeding atlas for Britain and Ireland and for the start of a number of the more important surveys were in 1970, it was decided to obtain information from 1970 to the most recently available data (1999 in this report). Hence the index is based on breeding bird populations for the period 1970-99. Approximately 230 species bred in the UK during this period. Data were available for 219 species. These data come from a wide variety of sources (Tab. 2). Wherever possible, an annual measure of population size (either absolute, *e.g*. pairs, or relative, *e.g*. an index) for 1970-99 was sought. There were eight main data sources:

#### *2.1.1. Common Birds Census (CBC) and Waterways Bird Survey (WBS)*

CBC and WBS are long-running mapping surveys of breeding birds (Marchant *et al*. 1990, Crick *et al*. 1998, Baillie *et al*. 2001). CBC indices were calculated for each year for 69 species using a general additive model (GAM) with degrees of freedom set to the full span of years in each data set. This is equivalent to a log-linear regression model with a full annual effect (*ie.* without smoothing). Indices were generated using data from all CBC plot types (*ie.* farmland, woodland and special plots; see Marchant *et al*. 1990). CBC data were available mostly for the period 1970-99, although for House Sparrow *Passer domesticus* they were available only from 1975. Data from the Waterways

Bird Survey (WBS) - (the riparian equivalent of the CBC) were calculated in exactly the same way for four specialist waterside species; Common Kingfisher *Alcedo atthis*, Dipper *Cinclus cinclus*, Common Sandpiper *Actitis hypoleucos* and Grey Wagtail *Motacilla cinerea*, and began in 1975. Indices were not calculated if sample sizes had fewer than twenty plots in more than half the years with data.

### *2.1.2. Rare Breeding Bird Panel (RBBP)*

The RBBP reports several population estimates for each species. The lowest is based on proven breeding pairs, the highest is the maximum total number of pairs. Because proof of breeding is difficult to obtain for many species, the latter is more likely to reflect the true breeding population and is used in creating indices. The run of RBBP data covers the period 1973-98, though with some exceptions. For example, RBBP only included a few species in their reports some years after the instigation of the scheme (*e.g*. Common Quail *Coturnix coturnix*). For some, the panel ceased to report national populations during the time period (*e.g.* Common Goldeneye *Bucephala clangula*). Occasional years of data are missed for some species (*e.g.* Snow Bunting *Plectrophenax nivalis*, Black Redstart *Phoenicurus ochruros* and Marsh Warbler *Acrocephalus palustris*). For a few species, such as Cirl Bunting *Emberiza cirlus* and Dartford Warbler *Sylvia undata*, data from the panel are enhanced by full national surveys at decadal intervals. These are listed as RBBP/SURVEY in Tab. 2.

Since the monitoring of most rare breeding birds by the RBBP began in 1973, the indicator for rare breeding species was started at, and indexed to that year.

### *2.1.3. Seabird monitoring*

Seabirds are monitored by two separate schemes. The Seabird Colony Register (SCR) is a complete census of British and Irish seabirds every 15 years. In practice this has been in 1969-70 (Cramp *et al*. 1974) and 1985-87 (Lloyd *et al*. 1991). The Seabird Monitoring Programme (SMP) has counted a sample of plots throughout Britain and Ireland since 1986. For most seabirds therefore, population sizes are known for the two complete censuses, and trends are known for a number from 1986 onwards. Unfortunately, truly national post-1986 trends are available only for a small number of species (there are many regional trends). For Common Guillemot *Uria aalge*, Northern Fulmar *Fulmarus glacialis* and Sandwich Tern *Sterna sandvicensis*, a chain index was produced from 1986 to 1999 (see Upton *et al*. 2000). These species are listed as SCR/SMP in Tab. 2. Annual trend data were available for Little (*Sterna albifrons*) and Roseate Terns (*S. dougallii*). For some species (*e.g.* skuas), full national surveys have been undertaken since 1985-87. Such species are listed as SCR/SURVEY in Tab. 2.

### *2.1.4. Wetland Birds Survey (WeBS)*

For a small number of waterfowl, the best information on annual breeding population levels is available from the WeBS scheme (see Pollitt *et al*. 2000). Although this monitors mainly the non-breeding population, the WeBS trend can be taken as the breeding trend for sedentary species; ie those whose UK wintering population is made up solely of UK breeding birds. Such species were, *e.g.* Mute Swan *Cygnus olor* and Ruddy Duck *Oxyura* *jamaicensis*. Although year-to-year variations in trend will also be related to productivity in the previous breeding season, these are small compared to the overall trend. WeBS produces indices for winter months, which span the end of one year and the beginning of the next. The winter 1970/71 index was taken as the 1970 breeding season value, 1971/72 taken as 1971 value, and so on. The indices were generated using the Underhill method (Underhill 1989, Underhill & Prys-Jones 1994), with 1970 set to an index of 100.

### *2.1.5. Single-species survey data*

A number of species, though not monitored annually, are monitored intermittently on longer time scales; most commonly every ten years at a national scale. In recent years, much of this has been undertaken within the Statutory Conservation Agencies/RSPB annual breeding birds scheme (SCARABBS), although other organisations have also been involved. Such species are listed as SURVEY in Tab. 2.

### *2.1.6. Other population monitoring data*

Information on trends for a variety of other species was extracted from the scientific literature (OTHER); for Red Grouse (*Lagopus lagopus scoticus*) gamebags were used as the best index of the species population trend.

### *2.1.7. Distributional data*

For a number of species, some 42 out of the total of 219 (=19%) there were no data available on population size during the time period. For these species a change in range, rather than population, over a twenty-year period was used. These data were obtained by comparing the results of the breeding atlases of 1968-72 (Sharrock 1976) and 1988-91 (Gibbons *et al*. 1993). Data on population trends (rather than changes in range) were always used wherever available, even if they were for a shorter time period than that spanned by the atlases (Red-throated [*Gavia stellata*] and Black-throated [*G. arctica*] Divers). Wherever a population or range estimate was collected from a survey spanning more than one year, the value was allocated to the middle year(s) of the range of survey years. For example, values from the 1968-72 atlas were allocated to its mid year, 1970, while data from the SCR collected during 1985-87 were allocated to 1986.

The geographical scope of the data for each species is summarised in Tab. 3. In most cases (86%), the data are of change in population or range for the UK. This is because most of the major schemes (*e.g.* CBC and RBBP) cover the UK. In practice, some of these schemes yield trends that may be a biased representation of the true UK trends, largely because some have no formal sampling design. Data for half

Tab. 3. Geographical scope of the species data.

| Geographical scope | No of spp | Notes |
|---|---|---|
| UK | 188 | *e.g.* CBC, WBS and RBBP data |
| GB | 16 | *e.g.* WeBS wildfowl indices |
| UK coast | 5 | Only coastal part of seabird populations monitored |
| Northern Isles | 3 | Skuas and Arctic Tern *Sterna paradisaea*; bulk of populations are in the Northern Isles |
| Shetland | 2 | Whimbrel *Numenius phaeopus* and Red-throated diver |
| Other | 5 | Various |

of the remaining species are representative of Great Britain (GB) rather than UK (GB plus Northern Ireland); thus for example the WeBS trends used for a few wildfowl are indices for GB not UK. The remaining species data are from yet more restricted geographical scales. However, in all of these cases, the bulk of the UK population for that species lies within these areas. Thus, for example, for five species of seabird the UK coastal population is monitored, even though a small part of the population may nest inland.

### 2.2 Dealing with missing values

Ideally, one would have measures of population (or failing that, range) for all 219 species for each of the 30 years, 1970-99. In practice, this was not the case and there were many missing species-year values. These missing values were either of data that has never existed, or which have been collected but not reported at the time of the analysis. Wherever possible missing values were estimated by interpolation (ie in years between known values) or by extrapolation (ie in years beyond known values) in the following manner.

To interpolate missing values a constant annual rate of change (C) in between the intermittent surveys was calculated as:

$$C=(value_n/value_1)^{1/(n-1)}$$

Where: $Value_n$=value (e.g. population size or index) in $yr_n$, and $value_1$=value in $yr_1$.

Knowing C and $value_1$, it was possible to estimate the values for $yr_2$, $yr_3$, $yr_4$ etc up to $yr_{n-1}$. For species with several intermittent surveys, C was estimated for each intervening time period separately. The approach taken to deal with missing values at the beginning and ends of data series was to extrapolate forwards or backwards based on the species trend over the previous or following periods. No data were extrapolated (forwards or backwards) over more than a nine-year period. This period is almost certainly too long (see Discussion) and in subsequent versions of the indicator the period is likely to be reduced. Instead, alternative data sources will be sought, or the species may be excluded from the indicator.

Extrapolations were either from intermittent surveys or annual monitoring data. The method of extrapolation was subtly different for these two sorts of data. An explanation of forwards extrapolation is given here, but the principle is the same for backwards extrapolation.

For intermittent surveys, the interpolation formula (above) was used for forward extrapolation beyond the last survey. Where there were several intermittent surveys, the most contemporary value of C was used. The manner in which missing values were extrapolated for annual data was similar to the forward extrapolation from intermittent surveys, but with C calculated from the mean of the first and last three years of data in the monitoring string.

One drawback with this approach is that it assumes a linear change from beginning to end of the data string, and this cannot always be justified. Annual monitoring data were only rarely extrapolated forwards by more than two years.

### 2.3. Calculation of the mean index

Since population size is measured in a variety of units (e.g. pairs or indices, often with different base years for indices), it is necessary to standardise all figures to a base year. We chose to use 1970 (the first year in the index) as the base year. This may give the impression that the 1970 value was some kind of target to be regained, particularly with an index that declines from 1970, but this was not the intention. Species for which no data for 1970 were available or where they cannot be extrapolated from later years (because of incompatible survey techniques, for example) were excluded.

The mean index was calculated as an average index of population trend taken across species (or various groupings). One cannot take a simple arithmetic average of indices. Instead, for each year separately, the log of each species index value was taken, this was then averaged across species and the exponential of the result calculated. Hence, each indicator is simply the average population trend of the species that it includes.

**2.4. Groupings of species prior to index calculation**

Each species was classified in three separate ways, by native or introduced status, by habitat and by abundance class. These classifications allowed the calculation of across-species indices for different groupings. Each species was categorised as native or introduced/feral following the definitions used by Gibbons *et al*. (1993). Re-introduced, or part-re-introduced species (Capercaillie *Tetrao urogallus*, Osprey *Pandion haliaetus* and Red Kite *Milvus milvus*) were included as native species.

Each species was allocated to one of seven habitat categories. These categories, which reflect the main habitat used for foraging during the breeding season, were: coastal, farmland, woodland, wetland, urban, upland and 'not classified'. The classification follows Gibbons *et al*. (1993), parts of which were taken from Ratcliffe (1990, for uplands) and from Fuller (1994, for woodland). Twenty additional species were allocated to their preferred habitat because they were too rare or had too restricted a distribution to be categorised by Gibbons *et al*. (1993).

In this situation, a species can be included in only one habitat, even though it may occur in many different habitats. There is no reason why species could not be included in the different habitats they occupy (with an indicator for each), but this would slightly alter the nature of the indicator.

Each species was classified as rare (<500 breeding pairs in UK) or not rare (>500 breeding pairs) at the time of the most recent population estimate included in Stone *et al*. (1997). For a few species, it was necessary to convert into pairs the unit in which population size was reported (*e.g.* adults or individuals), following Heath *et al*. (2000).

**2.4. Rare bird indicator**

The rare bird indicator follows the methodology of the headline indicator but there are important differences due to data availability and quality. Thus, the index runs 1973-1998, the period for which these data were available. In calculating a mean index, a five-year running mean was used as the species-year value, instead of the real count value. This not only allowed smoothing of the sometimes large fluctuations in yearly counts of some very scarce species (through variations in observer effort and the difficulties inherent in surveying scarce animals), but also the inclusion of some very scarce, colonising or

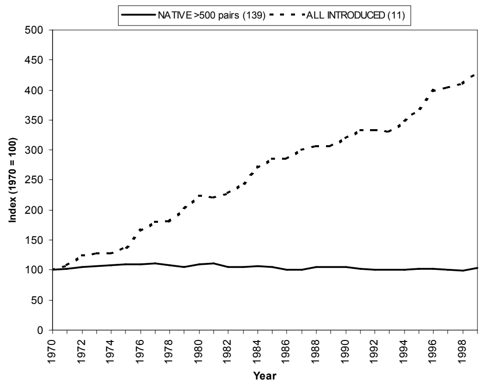



Fig. 1. UK Headline wild bird indicator for 139 common (more than 500 pairs) native species and indicator for 11 introduced species.

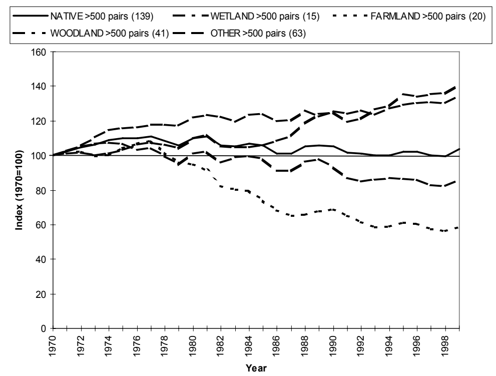



Fig. 2. UK Headline wild bird indicator for 139 common (more than 500 pairs native species and indicators for species of woodland, wetland, farmland and for unclassified species with populations greater than 500 pairs.

declining species that would have otherwise been excluded from the mean index. Species with an index of zero in any year were set to an arbitrary index value of 1 (van Strien, pers comm).

## 3. Results

Annual population indices (both real and estimated values) were available for 198 species for the period 1970-99. Of these, 11 were of introduced or feral origin and their overall populations have increased strongly (Fig. 1). Populations of some other groups of species, most notably wetland birds, increased during 1970-99 (Fig. 2). Among the remaining 187 native species, 42 had populations of fewer than 500 pairs. Populations of these rare species have increased substantially, rising by over 260% between 1973 and 1998, as shown in the separate rare species indicator (Fig. 3). Rarities were excluded from the final headline indicator because their population trends were not representative of the wider environment, most having increased because of direct conservation action. The final headline indicator was thus based on trends of the remaining 139 common native species, indices being produced for all 139 species combined, and for farmland and woodland birds (subsets of the 139) separately (Fig. 4). While the overall line has remained relatively constant, the woodland and farmland indices have fallen by approximately 20 and 40% respectively since the mid-1970s. Farmland and woodland account for about 85% of the UK land surface and are home to many of the UK's most abundant species. Declines of species in these habitats are probably a sign of general environmental change or deterioration.

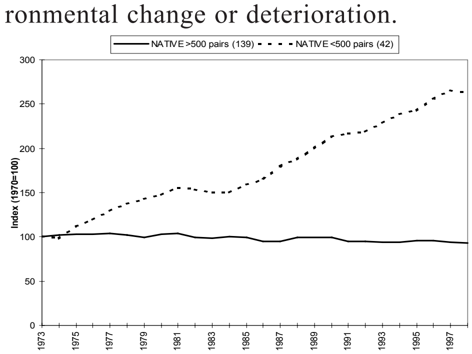



Fig. 3. UK Headline indicators for 139 common (more than 500 pairs) native species and 42 rare (fewer than 500 pairs) species.

It has also been possible to produce headline and rare species indicators for specific regions and countries within the UK, following methods similar to those used above. These are currently under development, but examples are given in Figs. 5 and 6. It can be seen that there are considerable differences in the indicator trends in different regions and habitats, which are only partly due to the differences in species composition of the regional avifauna.

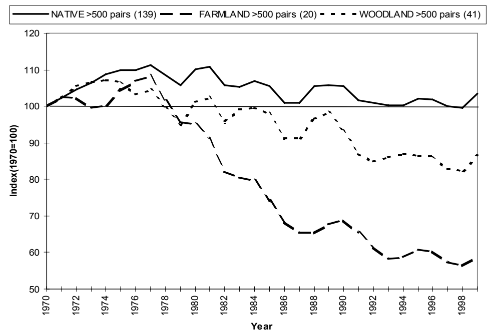



Fig. 4. UK Headline wild bird indicators for 139 common species and for common woodland and farmland species, as accepted by the UK Government as one of its 15 Quality of Life Indicators.

## 4. Discussion

### 4.1. General remarks

Here we describe a new method for producing wildlife indicators based on an average index across all species. A version of this mean index, representing the commoner native bird species (Fig. 4) has been adopted by the UK Government as one of its 15 headline indicators, the so-called Quality of Life Indicators, out of a set of 150 core indicators of sustainable development (Anon 1998, 1999). It shows unequivocally, that on average, common birds of both farmland and woodland are in sharp decline. It is recognised that such an index has resonance with policy makers, politicians and the public alike. The UK Government is committed to publishing annual updates of the headline indicator, its goal being to reverse the long-term trends. Furthermore, the Ministry for Agriculture Fisheries and Food has pledged to reverse the decline of farmland birds by 2020, using the headline indicator to measure their progress. There is mounting evidence that farmland birds are threatened in the UK (Marchant *et al*. 1990, Gibbons *et al*. 1993, Marchant and Gregory 1994, Fuller *et al*. 1995, Baillie *et al*. 1997, Siriwardena *et al*. 1998) and in Europe (van Strien 1997), and that the driver of these changes

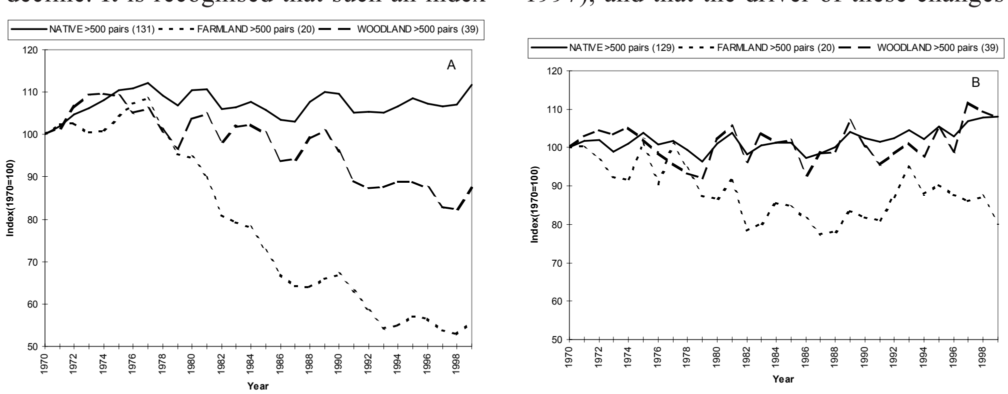



Fig. 5. Examples of regional Headline indicators currently under development (a) England, (b) Scotland.

Tab. 4. A comparison of the properties of different wildlife indicators.

| **(a) A mean index approach** | **(b) AMEOBA approach** | **(c) Red List Index** | **(d) Ecological Capital Index** |
|---|---|---|---|
| All widespread species are included | Indicator species are selected | Rare species included | Species indicative of habitats are chosen |
| All species are weighted equally | All species are weighted equally | Influenced by changes in status of species of high conservation concern | All the included species are weighted equally |
| Underlying model is simple | Underlying model is simple | Underlying model is simple | Underlying model is complex |
| There is no reference state/period | A reference state/period must be chosen | A reference state/period must be defined | A reference state/period must be defined |
| Require high quality data | Require high quality data | Require lower quality data i.e. categorical data on rare birds | Require high quality data for chosen species |
| All species need to be monitored | Indicator species need to be monitored | Rare species need to be monitored | Indicator species need to be monitored |
| Sensitive to change | Sensitive to change | Relatively insensitive to change | Sensitive to change |

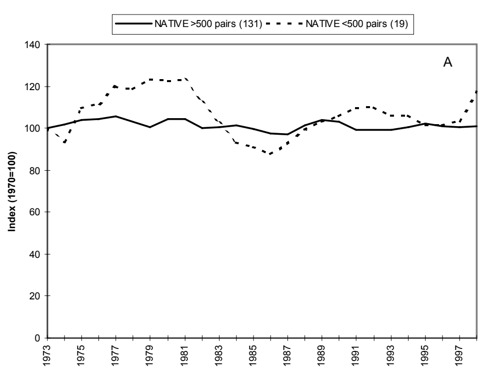
is agricultural intensification (Krebs *et al*. 1999, Donald *et al*. 2001).

Recent work has extended the general methodology presented here to examine regional variation in common and rare breeding bird populations within the UK, and to produce regional headline and habitat-based indicators (Figs 5 and 6.). The methodologies of the UK-wide survey schemes from which constituent data are drawn lend themselves to collation of data on a regional basis. Regional wild bird indicators are currently being developed as one type of a number of indicators of regional sustainability in conjunction with the UK Government.

Data on non-breeding bird populations were available for many (but not all) species, but they were not incorporated in the index. These may be incorporated at a future date, with the possibility of including an indicator of wintering bird populations in the UK, because the UK is globally important as a wintering site for many wildfowl and wader species.

## 4.2. Conceptual issues

The wild bird indicator is the average trend of a group of species found in a particular country, region or habitat, and the degree to which this indicates changes in



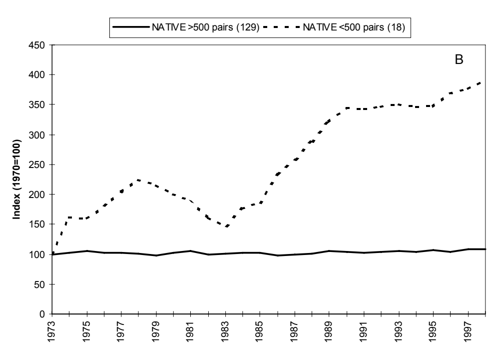




Fig. 6. Examples of regional rare species indicators currently under development (a) England, (b) Scotland.

the landscape or biodiversity in general remains open to question. In the case of UK farmland, declines in bird populations have been mirrored by declines in populations of many specialised invertebrates and plants, declines driven mostly by similar changes in land use (Donald 1998, Sotherton & Self 2000). Whether birds can act as bio-indicators in other ecosystems and in other situations is less clear. In some, perhaps rare, cases, population gains among birds could reflect habitat degradation *e.g.* mild eutrophication, rather than any genuine improvement in habitat quality. This reinforces the need to be cautious in promoting birds as indicators of other wildlife.

**4.3. Statistical issues**

The use of atlas range change in the place of abundance data (for 42 out of 139 species in the headline indicator) is contentious and its use in future breeding bird indicators is under review. Range change, based on two widely spaced surveys, is a relatively insensitive measure of trends in bird populations. First, the use of atlas data assumes that changes in range and abundance are analogous. This may not always be true and the degree to which the two are linked may be species specific. Second, extrapolation of these data assumes a linear consistent change over the entire period, including after 1990. In the absence of any evidence that this assumption is justified, we should be aware that these extrapolations may differ considerably from actual changes. It seems likely that the atlas information will not be used in future updates of the headline indicators.

In this work, no assessment of the precision of the indicator has been made. Some measure of statistical confidence would be desirable if trends shown by the indicator are to be ascribed to real processes, rather than to chance fluctuations. When dealing with single species population indices derived by GAMs or similar, this can be achieved by calculating confidence intervals by bootstrapping on survey sites (Buckland *et al*. 1992, Siriwardena *et al*. 1998). The bootstrapping approach could also be adapted for use in a multi-species indicator. It is not possible to estimate the precision of data from some of these sources, and hence the average trend may incorporate these ‘unknown’ errors. However, it may be possible to use analytical solutions to approximate errors of the mean index (van Strien, pers comm).

**4.4. Alternative indicator models**

The development of sustainable indicators in the UK parallels work elsewhere. In the Netherlands, for example, three separate indices have been developed, termed the AMOEBA approach (Ten Brink 1991), the Red List Index and the Ecological Capital Index (van Strien 1997, 1999). The general properties of these indicators (and the mean index) are given in Tab.4.

The AMOEBA approach is an innovative method that compares the status of a number of species at some recent point in time with a previous reference point, the latter being chosen to represent an ideal state (Ten Brink 1991, Ten Brink *et al*. 1991). This approach can also compare two systems separated in space where, again, one is chosen to represent an idealised state. The index can include a range of species, although there is some preselection. The visual presentation of the indicator is one of its key characteristics and was developed with non-specialists in mind. The outputs show the difference between the present and the reference situation, and their amoeba-like form gives the indicator its name. A further product of this diagram is termed an 'ecological Dow Jones Index' that is the summed numerical difference between the reference points and the observed data for all the species. The smaller the difference, the closer the system is to a desirable state. This approach can be criticised because different taxa are included with equal weighting, although they may have different values to some users and there is subjectivity in choosing the species and the reference condition.

In the Red List Index, the rarity of a species is classified into one of five groupings, which have different associated scores linked to range or numbers at several time points. The scores are then summed across species for each period and expressed in relation to the reference period. Van Strien (1997, 1999) was able to calculate indices for eight taxa, and in all cases but one, the index showed biodiversity to have declined in the Netherlands since 1900. Curiously, the exception was birds; overall, rare breeding birds had increased. This result thus parallels our own findings for the UK (Fig. 3). Rare birds have increased in both countries because of concerted conservation actions to protect and enhance the species and their habitats. Clearly, the Red List index is not designed to deal with common species, rather it is designed for use alongside the Ecological Capital Index (see below). A further criticism is that the classification of species into broad classes of rarity may be too crude, and so species can move between classes only rather slowly.

The Ecological Capital Index (ECI) is arguably the most sophisticated of the methods considered here. This habitat-based approach combines the quality and quantity of a habitat into a single figure. Quality is taken to be the density of a number of habitat-specific species, and quantity is the area of that habitat. Both rare and common species can be included and their contemporary densities are contrasted with a reference situation in the past. Habitat quantity comes from land cover statistics and is expressed in relation to the reference period. The ECI is the product of quality and quantity. Using birds as an exemplar taxon, van Strien (1997, 1999) showed a decline in habitat quality and quantity in the Netherlands, using the 1950s as the reference period. Overall, farmland and heathland habitats had deteriorated to the greatest extent. This basic framework has also been used with slight modification in the Natural Capital Index that is again based on concepts of ecosystem quality and quantity (Ten Brink 1997). One of the difficulties of this approach is that it concatenates two fundamentally different but related processes; the loss of habitat area and the loss of biodiversity inhabiting that habitat. One could have a situation where the area of a habitat declined rapidly but the biodiversity of the remaining patches was unaltered, or a situation where the habitat area remained constant but the biodiversity declined rapidly, yet both might have the same ECI. Disaggregating the index into its component parts provides better understanding of the ECI. As van Strien is careful to stress, there are two main practical difficulties; they are the choice of the reference period and the selection of the habitat-specific species. While the selection procedures have been based on expert advice, it is still arguable whether they can be considered strictly objective. The choice of species is akin to defining 'keystone species' (Paine 1969), a concept that is generally considered unworkable (Scott Mills *et al*. 1993). However, by taking a relatively wide group of species for each habitat, the amount of subjectivity is minimised. Future editions of the ECI are likely to take a broader group of species, thus increasing its similarity with the UK index (van Strien pers comm).

One of the main differences between the mean index approach and the other biodiversity indicators (Tab. 4) is that the former treats all species equally, regardless of conservation status, and does not include conservation targets. This may be seen as a strength or a weakness. On the positive side, there is no subjectivity in the choice of species to be included or the relative importance they may have because it covers all species for which data are available. However, since all species are weighted equally, 'desirable' rare or vulnerable species are treated equally with 'less desirable' common, or even pest species. This reinforces the point that indicator information needs careful thought and interpretation. Disaggregating the trends is an important step in understanding the underlying patterns. The method we present allows the simple presentation of large amounts of ecological data, making it available to a diverse non-expert audience. While our method has some inherent limitations (and should be regarded as a simplistic summary of a complex situation), it has proved to be an effective tool in communicating information about biodiversity to the public, policy makers and to Government in the UK.

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**Ornis Hungarica 12-13: 25-31. 2003**

# Developing a general conceptual framework for avian conservation science

J. R. Sauer

Sauer, J. R. 2003. Developing a general conceptual framework for avian conservation science. – Ornis Hung. 12-13: 25-31.

Avian conservation science in North America has produced a variety of monitoring programs designed to provide information on population status of birds. Waterfowl surveys provide population estimates for breeding ducks over most of the continent, the North American Breeding Bird Survey (BBS) provides indexes to population change for >400 breeding bird species, and many other surveys exist that index bird populations at a variety of scales and seasons. However, many fundamental questions about bird population change remain unanswered. I suggest that analyses of monitoring data provide limited understanding of causes of population change, and that the declining species paradigm (Caughley 1994) is sometimes an inefficient approach to increasing our understanding of causes of population change. In North America, the North American Bird Conservation Initiative (NABCI) provides an opportunity to implement alternative approaches that use management, modeling of population responses to management, and monitoring in combination to increase our understanding of bird populations. In adaptive resources management, modeling provides predictions about consequences of management, and monitoring data allow us to assess the population consequences of management. In this framework, alternative hypotheses about response of populations to management can be evaluated by formulating a series of models with differing structure, and management and monitoring provide information about which model best predicts population response.

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## 1. Introduction

The North American Bird Conservation Initiative (NABCI) is a collaboration among bird management plans in North America (For more information, see their website at http://www.nabci.org/cec/about_frame.htm). It includes a variety of partnerships, including government and private organizations, and is international in scope. Because it incorporates existing conservation activities such as Partners in Flight the North American Waterfowl Management Plan, the Shorebird Management Plan, and the Waterbird Management Plan, an enormous amount of effort is devoted to discussion of development of monitoring, management, and research activities for birds. Avian research efforts associated with NABCI focuses on 5 primary topics: monitoring, integrated modeling/analysis, decision support, adaptive management, and information management (http://www.pwrc.usgs.gov/nacwcp/nabcidr.pdf). NABCI provides a unique opportunity to consider the relative role of these activities in increasing our understanding of the causes of bird population change.

## 2. Monitoring

### 2.1. History

Monitoring has long been a primary focus of bird management. Documenting changes in populations over space and time is fundamental to any conservation or management activity, and the notion of tracking population response to management is well established. Caughley (1994) described 2 models for conservation. The "endangered species" paradigm is applied to small populations at risk of extinction. For these species, genetics concerns are important considerations, and population dynamics modeling such as population viability analyses are often conducted. The "declining species" paradigm is the alternative strategy of monitoring populations to identify species that are declining in population, and then conducting research to identify causes. Migratory bird conservation activities in North America historically have relied upon of the declining species paradigm, in that effort is first directed in developing monitoring to identify population declines. Once these changes are identified, conservation actions are developed to prevent species from further declines (USFWS 2000).

### 2.2. Shortcomings

Unfortunately, research does not always provide coherent answers to managers. A variety of North American species have shown long-term population declines, but the causes of these declines remain obscure even though many research studies have attempted to identify causes. Examples of these taxa with uncertain causal factors for declines include Black Ducks *Anas rubripes*, *e.g.* The Black Duck Joint Venture Strategic Plan (http://www.pwrc.nbs.gov/bdjv/bdjvstpl.htm), Neotropical migrant birds, and grassland-breeding birds (*e.g.* Peterjohn & Sauer 1999). Many important questions are still unresolved for most species, including such fundamental questions as:

1. Relative importance of wintering ground and breeding ground in influencing population change.
2. Relative influence of environmental features on survival and productivity.
3. Influence of harvest on bird populations.
4. Influence of local habitat management on bird populations.
5. Influence of habitat management at a landscape scale on bird populations.

These questions are still controversial for several reasons. The scale of some questions is beyond our current resources or tools. Banding is an insufficient tool for addressing many demographic issues and other complicated questions of movement rates among breeding and wintering sites of migratory birds. Often, the scale of experiments is local, and extrapolation to regional populations is uncertain. Estimation of survival rates from radio-tagged birds and local productivity analyses are examples of local studies that are often difficult to extrapolate to a regional scale. Coordinated experiments with appropriate sampling frames that provide inference to regions using these intensive tools are still very rare. Models of bird-habitat relationships are similarly limited in scale. Generally, to evaluate regional-scale hypotheses, we rely on association analyses where 'treatments' (*e.g.* habitat manipulations, harvest regulations) are not experimentally applied. Unfortunately, it is generally difficult to establish causality in these association analyses.

### 2.3. The debate

This uncertainty on causes of observed population changes has led to introspection about the process of management and the role of managers and researchers in bird conservation. Subtle differences of opinion exist about how information is acquired and used, and whether monitoring should provide general information on population status or be an active tool with specific goals. Tools such as decision support systems and geographic information systems provide new opportunities for managers to make monitoring an explicit part of management, with clearly defined goals. These tools also provide the opportunity to use models to predict consequences of management on bird populations, and provide new goals for monitoring in evaluating predictions from models. NABCI provides an opportunity for researchers and monitoring specialists to evaluate their role in increasing our understanding of bird population dynamics.

### 2.4. Limitations of the declining species paradigm

This paradigm is the prevailing idea for much of bird conservation. In the declining species paradigm, bird conservation has 2 phases: observation of population change, and then research into causes of declines. Unfortunately, this approach is inefficient, as observing declines does not lead to understanding of causes of declines. Because observation of declines tends to trigger simultaneously both management and research, it encourages action to mitigate problems at the same time as research is in progress. It justifies monitoring for monitoring's sake, rather than considering it as part of management. When evaluation of causes is distinct from management, there is no impetus to think in an integrated manner about the roles of research, monitoring and management.

Management of populations is extremely difficult when decisions must be made based only on monitoring data. For example, it is impossible to interpret the biological significance of most population declines estimated from monitoring programs. Often, arbitrary population changes are set as standards, and estimated population changes that exceed these thresholds are considered for additional management and research. However, without additional information on the context of the population change estimates, most of these thresholds are meaningless. Occasionally, causes of population change are obvious, and can be evaluated by association analyses of monitoring data. Often, however, changes are subtle rather than obvious, and managers cannot determine the context for the observed population change.

This lack of generally accepted standards for defining population declines is a complication in any species prioritization process (*e.g.* Carter *et al.* 2000).

An additional complication associated with migratory bird conservation is that, in the past, managers have not received clearly defined management options, and their ability to predict the consequences of their management has been poor. Management options have been limited, and in North America the emphasis on management of harvested species reflects the notion that for these species an obvious management tool exists. For land managers at local and regional scales, the management options for migratory birds have been even more limited, because little information exists on management of habitats for migratory birds. Local management has relied on simple bird habitat models that generally are not based on experimental studies of the relationships of population change and habitat change. Defining management options and implementing reasonable monitoring systems at these scales is a fairly recent innovation.

## 3. Escaping the declining species paradigm

### 3.1. Defining scales and systems for management

NABCI has concentrated conservation efforts on habitat management at local and regional geographic scales. In particular, Bird Conservation Regions (BCRs, Fig. 1) have been developed to provide a common geographic framework for conservation in North America. Within these regions, management plans define priority species and plan conservation activities. Clearly, conservation activities include management of habitats to modify suitability for priority species. This definition of spatial scales and areas of conservation interest is accompanied by development of geographic information that can be used by managers to assess available habitats. These tools permit emphasis by managers on systems and scales of interest and on options for management that can be rigorously defined in terms of geographic models. Local land managers can evaluate the consequences of changing land use on parts of their areas, and regional landscape managers can evaluate changing land-use patterns at the regional scale.

### 3.2. New information sources help in decision support

Remote-sensed data and geographic information systems provide a variety of new tools to describe habitats and bird populations for local and regional management. Managers can use these tools to define habitats in areas to be managed, developed predictive models in the geographic context, and to describe alternative management scenarios. These decision support tools can be used provide quantitative information on local and regional landscapes and habitats, but only recently have managers begun to gain access to these sophisticated tools. A great deal of additional work is needed to develop tools that allow managers to use decision support tools effectively in management.

### 3.3. New notions on use of management as a tool for increasing understanding of systems

Our limited understanding of causal factors influencing population change, and our limited abilities to develop appropriate experiments to evaluate factors influencing bird populations, have led to the idea that management often provides our best tool for learning about factors influencing population change. Historically, monitoring and management of harvested species such as waterfowl has provided data used in association analyses. Unfortunately, these association analyses only provide weak evidence of causes of population change. Adaptive management is an alternative approach that provides a coherent framework for assessing causality.

Adaptive management is a model-based approach to management that acknowledges uncertainty in our understanding of how management influences populations. In adaptive management, models are used to predict outcomes of management and to choose an appropriate management strategy. Management then occurs, and monitoring is used to assess the results of management. Monitoring results are compared to predictions of the models, and model selection for use in prediction is updated to reflect the new information on how management influenced the population. Later

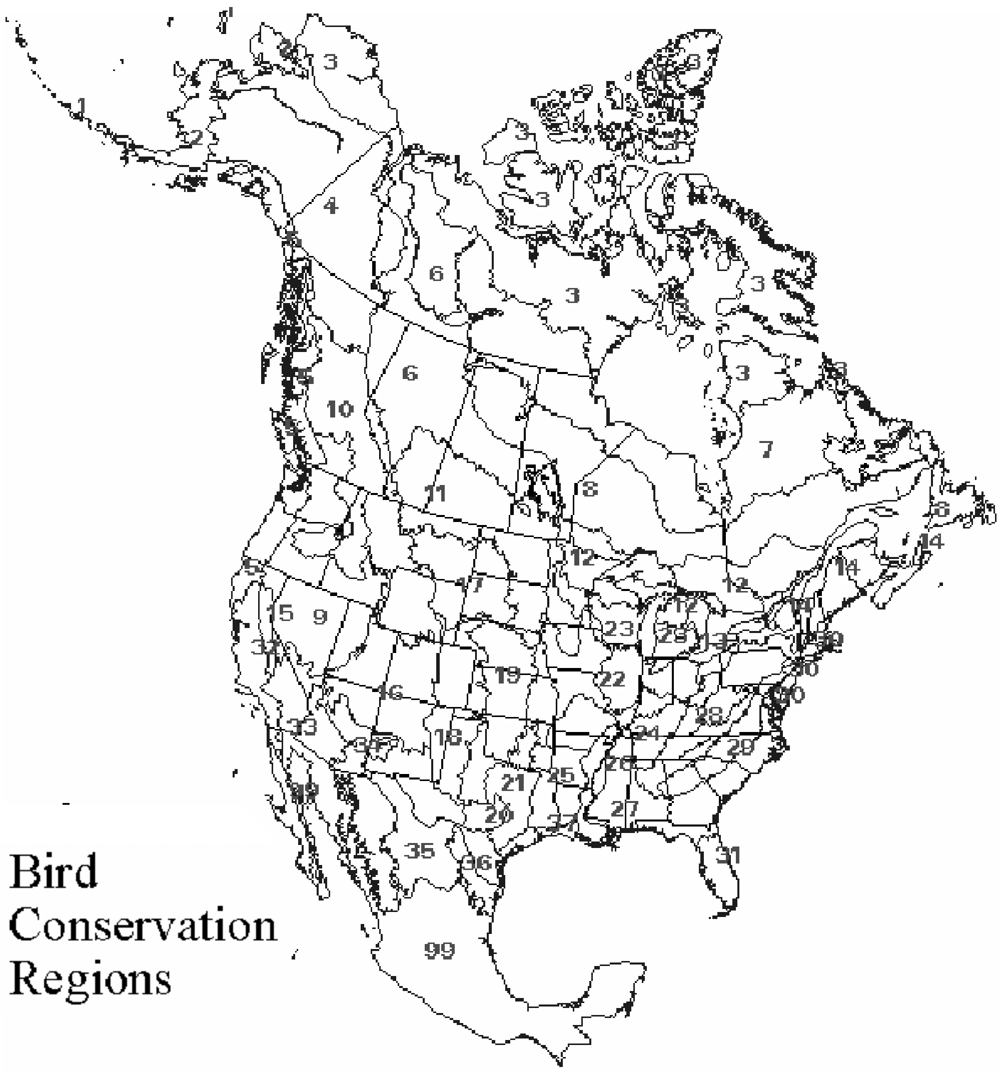


Fig. 1. Map of Bird Conservation Regions, as defined by the North American Bird Conservation Initiative (R. Johnson, United States Fish and Wildlife Service *pers comm*).

management subsequently uses the updated models for prediction and selection of the best management action. This approach is used in harvest management of selected species in North America (Williams & Johnson 1999). It provides a coherent framework for defining management goals, organizing research information into models, and applying the results to subsequent management decisions; monitoring is then implemented to assess the species population response to management.

### 3.4. Developing models of systems of interest

Adaptive management requires an ability to predict the consequences of management. This requires us to formalize our understanding of the system by developing predictive models about how management will influence population change. Models are supposed to explain essential elements of the system, incorporating both our knowledge of the system and the uncertainties associated with our knowledge. Although managers use many types of models at present, much of the present modeling is based on qualitative information that does not provide specificity for management. Now, new opportunities exist for development of quantitative models, because definitions of goals and scales of management provide an explicit context for developing models to describe the effects of management on birds. Furthermore, decision support tools provide additional structure by providing information on relevant habitat and environmental covariates for management.

### 3.5. Making models

To make a model, one must formally define the physical boundaries of the system. For bird conservation in North America, systems are frequently defined in terms of areas such as:

1. Refuges and surrounding landscapes.
2. National Parks.
3. Bird Conservation Regions.

For the system, it is necessary to define the state variables, the variables that are to be modeled (*e.g.* population size). It is essential to include in the model:

a. Exogenous variables: *i.e.* factors that influence population change but which cannot be controlled, such as weather and water levels.
b. Control variables; *i.e.* factors that influence population change that can be managed, such as harvest and habitat.

Finally, a transition equation must be developed that defines how variables interact over time to influence population change. Often, a transition equation is not known exactly, but we can define alternative possibilities in a series of models. In all modeling efforts, it is important to incorporate uncertainty of the estimates of these factors.

Modeling is an obvious component of any management of populations, and all migratory bird conservation fits implicitly into a system that could be modeled. Experimental work plays a large part in the development of models, model structure and it is also crucial in the estimation of components. Systems are never completely understood, but this uncertainty is implicit in both modeling and management. Our models change, either as knowledge expands through experimentation or upon examination of management results. Management actions followed by evaluation is the only possible method of increasing our understanding of many of our systems. Consequently, ties to management must be explicit in models.

## 4. The role of monitoring in adaptive management

Monitoring has a critical role in adaptive management in that it allows us to assess system status in the context of a model, providing a basis for assessing results of management. This role is fundamentally different from monitoring's role in the declining species paradigm, where often observation of pattern becomes disconnected from understanding of causes. Of course, the traditional roles of monitoring programs remain relevant in documenting patterns of bird population change and in bringing public attention to bird populations. However, it is important to recognize that monitoring data are not sufficient to address critical questions about causes of population change, and we presently rely too much on association analyses as surrogates for research.

### 4.1. Framework for avian conservation?

NABCI provides a possibility for getting away from the declining species paradigm, in which perception of interval-specific change drives management actions and in which qualitative notions of bird-habitat (or bird-harvest) associations are used to make management decisions. To develop a new framework for avian conservation, the essential requirements are:

1. Collaboration with managers in understanding:
   a. Systems of interest.
   b. Available (and needed) information.
   c. Management options.
2. Integration of information on systems through model development.
3. Experimental work to help us understand systems.
4. Development of alternative models when controversy exists about the effects of management.
5. Use of management as source of information on the validity of models through the use of adaptive management. Monitoring has a very focused role in assessing change in system status associated with management and hence in evaluating model predictions.

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**Ornis Hungarica 12-13: 33-40. 2003**

# Conservation of european farmland birds: abundance and species diversity

C. Stoate, M. Araújo and R. Borralho

Stoate, C., M. Araújo and R.Borralho. 2003. Conservation of european farmland birds: abundance and species diversity. – Ornis Hung. 12-13: 33-40.

Across much of Europe, farmland birds have declined more than those in other habitats; many of the most threatened birds are dependent on extensive farming systems. This paper describes two case studies in which bird abundance was monitored in relation to spatial and temporal differences in agricultural management, one in southern Portugal, the other in central England. In Portugal, bird abundance and species diversity were monitored in 1995 using transects in relation to three agricultural systems. Bird abundance and species diversity were both low in simple intensively managed farmland, and highest in extensively managed farmland incorporating agroforestry systems (Montado). However, species of greatest national and European conservation concern were most abundant in simple, open, extensively managed landscapes. These extensive systems are therefore important for species diversity at national and European scales. In England, a conventionally managed farming system was adapted to encourage gamebirds for shooting, and bird abundance was monitored annually. Transects were conducted within the study area from 1992 to 1998 and additional transects were conducted randomly in the surrounding farmland from 1995 to 1997. Bird abundance increased during the management period and was higher in the study area than in the surrounding area, especially for nationally declining species. However, there was little difference in species diversity across years or sites. Our monitoring demonstrates three important points: 1- Extensive farming systems play an important role in maintaining species diversity at national and European scales, even where abundance and diversity are low at the farm scale. 2- Abundance of nationally declining bird species can be restored rapidly, following population declines, if appropriate management systems are adopted. 3- Bird conservation can be accommodated within multifunctional land-use systems, including agricultural systems incorporating game management.

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## 1. Introduction

Farmland ecosystems have evolved through the development of agriculture, becoming almost uniquely characteristic plant and animal communities (Potts 1991). However, rapid changes in farming methods in the second half of the 20$^{th}$ century resulted in the partial collapse of this ecosystem (Potts 1997). Throughout most of Europe, farmland birds have declined as a result of simplification of farming systems and increased use of external inputs (Tucker & Heath 1994, Fuller 2000). 43% of bird species associated with arable habitats now have an unfavourable conservation status (Tucker 1999). Such responses to agricultural intensification are particularly well documented in northern Europe, but in southern Europe, abandonment of agricultural land has also caused severe declines in some bird populations (Beaufoy *et al*. 1994). Increasingly, farmland bird species are becoming an important focus for conservation policy (*e.g*. Tucker 1997, Swash *et al*. 2000) and are being used as indicators of wider ecological changes (Tucker 1999).

This paper reviews two studies of farmland bird communities in relation to farming systems in Portugal (Araújo *et al*. 1996) and England (Stoate & Szczur in press, Stoate in press). The Portuguese study compares bird abundance and species diversity between one intensive and two extensive farming systems within one year. The English study monitors bird abundance and species diversity over a seven-year period in which a conventional farming system was adapted to meet the ecological requirements of wild gamebirds. This game management system aims to adapt a modern farming system to provide some of the ecological conditions found in former extensively managed systems, but with minimal economic impact on the farm as a business. The monitoring aims to identify a potential role for such management in the conservation of declining farmland birds.

## 2. Study areas and methods

### 2.1. Portugal

The study area included parts or all of five administrative regions in Baixo Alentejo (Ferreira do Alentejo, Aljustrel, Castro Verde, Ourique and Almodôvar) and covered a total area of 155 000ha. Within this region, three land-use systems were recognised: intensive agriculture, extensive agriculture and Montado.

The intensive agriculture category is characterised by a greater frequency (>55%) of heavy soils, much of the area being irrigated. Wheat *Triticum aestivum* and barley *Hordeum distichum* are the main cereal crops and silage grass *Lolium sp*, sunflower *Helianthus annuus*, sugar beet *Beta vulgaris* and oilseed rape *Brassica napus* are also grown. Wheat yields are 2.5-3.5 tonnes/ha$^{-1}$ without irrigation but can be almost doubled with full irrigation (P. Eden *pers comm* 1998). There are short rotations with little or no fallow (*e.g*. sunflower - 1$^{st}$ cereal - 2$^{nd}$ cereal). This system requires frequent use of fertiliser (130 units $N_2$/ha$^{-1}$ (P. Eden *pers comm* 1998) and herbicides, relative to the other land-use categories. With the exception of some olive *Olea europea* groves, there is little tree cover.

The extensive agriculture category is characterised by thin soils and by the largest average farm size of the three categories. There is no irrigation and fallow area is relatively high. A typical rotation takes the form: plough fallow - 1$^{st}$ cereal - 2$^{nd}$ cereal - fallow - fallow, with fallow periods often lasting five years or more (Rio Carvalho *et al*. 1995). Wheat yields are 1.5-2.5 tonnes/ha$^{-1}$ with yields at the lower end of this range being more common. Triticale *Triticum aestivum* x *Secale cereale* and oats *Avena sativa* are frequently grown in the extensive category, and grazed or cut for silage. The incorporation of a fallow period into the rotation, and the relatively low potential yields are associated with considerably lower annual inputs than in the intensive category.

Montado (equivalent to the Spanish dehesa) is characterised by thin soils and tree cover, dominated by holm oak *Quercus rotundifolia* and cork oak *Q.*

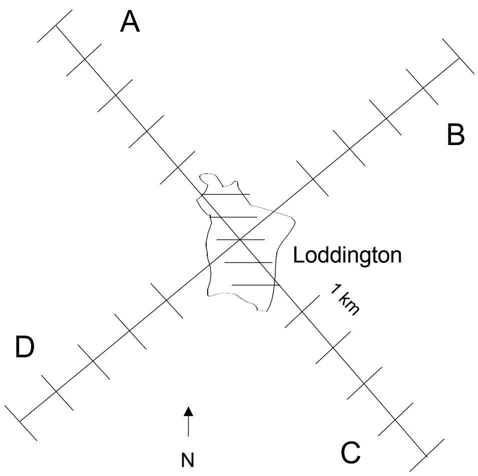


Fig. 1. Location of 1-km bird monitoring transects in relation to the Loddington farm boundary.

*suber*. Like the extensive category, there is no irrigation and the fallow area is high. A typical rotation is similar to that of the extensive category, although the fallow stage is often longer and it may include forage lupins *Lupinus luteus*. Sheep *Ovis aries*, cattle *Bos taurus* and pigs *Sus scrofa* are kept in all three land-use categories. Zero grazing is adopted on some farms in the intensive category but livestock normally graze fallows. More detailed information on these land use categories is provided by Stoate *et al*. (2000).

One hundred and fifteen 250m transects, starting at 1km grid intersections and stratified by land-use categories, were walked along a random bearing (Intensive: n=42, Extensive: n=42, Montado: n=31). Transect counts were conducted by a single observer in the first three hours after dawn of December 1994, 1995 and 1996, and April of 1996 and 1997. All adult passerines seen or heard, other than those flying across the count area, were recorded. For each category of land use, species were categorised into three levels of conservation concern (1 = rare, 2 = vulnerable, and 3 = other species thought to be declining but for which reliable data were not available) based on the criteria of SNPRCN (1991). In addition, total bird abundance and an overall Shannon-Wiener index of species diversity (Magurran 1988) were calculated for each land use category. Differences in bird abundance and species diversity between land-use categories were tested using ANOVA and LSD post-hoc tests (at $P<0.05$), using transects as sample units and log-transformed data.

## 2.2. England

The study area comprises approximately 150km$^2$ of mixed arable and livestock farms in Leicestershire, central England. The area consists of arable fields and grassland enclosed by hedges and there are numerous small woods. Soils are mainly heavy clay and the main crops are wheat, barley and oilseed rape. Within this area, transects (see below) were used to sample breeding abundance of birds in four discrete zones (Fig. 1). The main study area, at Loddington, is located at the centre of the wider study area and covers an area of 3.33km$^2$. The farm at Loddington is owned and managed as a research and demonstration farm by the Allerton Research and Educational Trust, the main incentive for environmental management being the management of wild pheasants *Phasianus colchicus* for shooting (Boatman & Brockless 1998).

Game management started in 1993, following a year of baseline monitoring. This management included thinning and replanting of woods and active management of hedges in order to increase the area of shrubby vegetation. Gamecrops were planted on 20m-wide mid-field and field-edge set-aside strips in order to provide invertebrate-rich foraging areas for gamebird broods in summer, and cover and seed food in winter. Pesticide use in cereal crops, especially on crop headlands, was restricted in order to increase arable invertebrate abundance (Sotherton 1991). Beetle banks and herbaceous strips in field boundaries were established to provide nesting cover for gamebirds and suitable summer and winter habitat for beneficial invertebrates (Rands 1987, Thomas *et al.* 1991). Legal control of potential nest predators was conducted from April to July each year (Tapper *et al*. 1996) and grain was provided by hand and from hoppers through the winter and early spring.

In the years 1992-1998, transect counts across all habitats at Loddington were used to provide an abundance index for each species. Transect counts were conducted by a single observer in fine weather in May and early June, in the first three hours after dawn. Four counts were conducted on foot each year at approximately fortnightly intervals. Transect routes totalled 11.5km and were constant between visits and years, incorporating well-defined habitats on each side of the transect line (ie the adjacent field and field boundary). All adult passerines seen or heard, other than those flying across the count area, were recorded.

In the years 1995-1997, separate transect counts were used to compare an index of breeding bird abundance at Loddington with that in the surrounding area. For this, five 1km long transects were conducted within Loddington and five transects were conducted at 1km intervals along each of four bearings radiating out from the centre of Loddington. The first bearing was selected at random, with subsequent bearings at 90°, 180°, and 270° to it. These formed four zones lacking wild game management for comparison with Loddington (Fig. 1). Each of the 25 transect counts was conducted once in May in the first three hours after dawn.

Data from the five transects in each area were pooled. Passerine species were divided into three categories: Biodiversity Action Plan (BAP) species (targeted for conservation action), other nationally declining species, and nationally stable or increasing species. In addition, a Shannon diversity index was calculated as a measure of species diversity (Magurran 1988). The index of total bird abundance (all birds counted), the species diversity, and the abundance indices for nationally declining and stable/increasing species at Loddington were all compared with the equivalent measures in the four zones in the surrounding area using log-transformed data and two-way ANOVA (zone×year) followed by contrast analysis (Loddington v average of zones A-D).

## 3. Results

### 3.1. Portugal

Overall bird abundance differed between all land-use categories ($F_{2,112}$=23.85, $P$<0.001), being significantly higher in Montado than in extensively and intensively managed farmland. Abundance of birds in all three categories of Portuguese conservation status differed significantly between land-use categories (Rare: $F_{2,112}$=18.02, $P$<0.001, Vulnerable: $F_{2,112}$=9.31, $P$<0.001, Other declining species: $F_{2,112}$=23.78, $P$<0.001); birds were

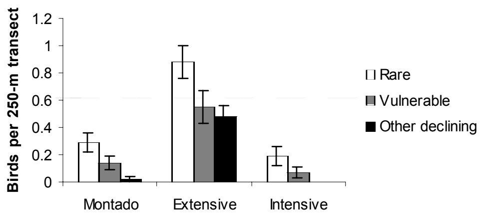


Fig. 2. Abundance of birds (mean ± se) of 'Rare', 'Vulnerable' and 'Other declining' conservationstatus in Portugal in relation to three Alentejo farming systems.

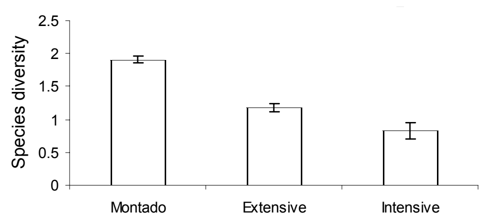


Fig. 3. Shannon-Wiener index of bird species diversity in relation to three Alentejo farming systems.

significantly more abundant in the extensive farming category than in intensive farming or Montado (Fig. 2). Species diversity also differed between land use categories ($F_{2,112}$=40.24, $P$<0.001), but was significantly higher in Montado than on extensively or intensively managed farmland (Fig. 3).

### 3.2. England

At Loddington, numbers of birds in the 'nationally declining species' category rose significantly over the seven-year period ($r_6$=0.87, $P$=0.01), the main increase occurring between 1992 and 1995, and were 129% higher in the 1995-1997 period than in 1992. Numbers of birds in the 'nationally stable and increasing species' category rose by 42% ($r_6$=0.66, n.s.) (Fig. 4). The Shannon Index of species diversity at Loddington increased from 1.10 in 1992 to an average of 1.16 in the 1995-1997 period, although this increase over the seven-year period was not significant ($r_6$=0.69, n.s.) (Fig. 4).

In the 1995-1997 period, there were no zone x year interactions in any of the variables examined. There was no difference in total bird abundance, bird abundance of nationally stable species, species richness or species diversity between Loddington and the average across the four zones in the surrounding area. There was a significant difference in BAP species abundance between Loddington and the average of zones A-D (contrast analysis $F_{1,9}$=7.52, $P$<0.05). Other nationally declining species were also significantly more abundant at Loddington than in the surrounding area ($F_{1,9}$=16.77, $P$<0.01; Fig. 5).

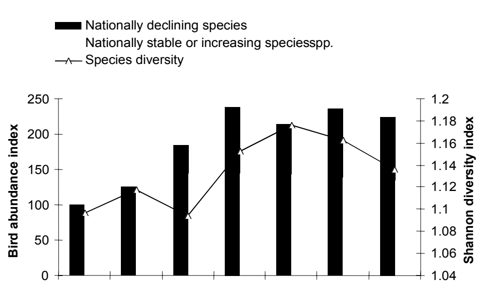


Fig. 4. Bird species diversity and relative abundance of 'nationally declining species', 'nationally stable or increasing species' at Loddington.

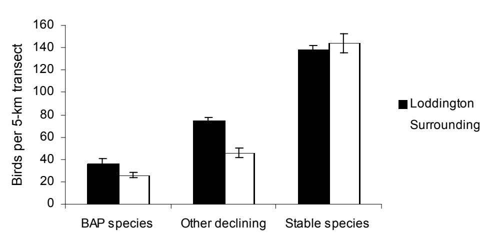


Fig. 5. Abundance (mean ± se) of Biodiversity Action Plan species, other nationally declining species, and nationally stable or increasing species at Loddington.

## 4. Discussion

Intensively managed farmland in both Portugal and England supports low species diversity and bird abundance. Although lost from England, extensively managed arable systems survive in Portugal where they are represented by such systems as Montado and the extensively managed arable steppe considered in this study. Montado supports both high bird abundance and high species diversity.

Higher species diversity within Montado may be explained by a combination of edge effects and the intermediate disturbance hypothesis. Montado comprises transitional systems between extensive arable steppe and forest and therefore it shares the bird communities of both systems. This is similar to Odum's (1971) 'edge effect' of increasing diversity as a result of the spatial overlap of species from neighbouring assemblages. Montado has intermediate frequency and intensity of disturbance, relative to intensive agricultural systems or old growth Mediterranean forest. Theory predicts higher mortality and lower productivity in highly disturbed areas (e.g. intensive farming), where diversity is low because populations of some species are unsustainable. At low levels of disturbance (e.g. old growth forests), mortality is reduced but diversity is low due to competitive exclusion, as the dominant species eliminate poorer competitors (for a review, see Huston 1994).

In this study, abundance of species of greatest conservation concern was low in Montado, relative to extensive arable steppe. Although Montado supports higher species diversity at the local scale, extensive arable steppes make an important contribution as habitats supporting globally threatened species such as Great Bustard *Otis tarda*, Lesser Kestrel *Falco naumanni* and other species of conservation concern within Portugal. In this study, most species observed within Montado are transitional species which also occur in other habitat types. These generalist species are not as threatened as those dependent on the specific conditions associated with extensive arable steppes. For example, within this habitat Moreira (1999) has shown that fallow area and structural diversity of vegetation influence bird abundance and species diversity. Maintenance of these habitats is therefore essential to the conservation of many farmland species. Nevertheless, even within more intensively managed areas of farmland, agricultural management could be adapted to meet conservation objectives, either by restoring traditional extensive management, or by introducing novel management practices that are designed specifically to meet the ecological requirements of nationally declining birds.

The results from England suggest that this approach can be successful. In this case, the incorporation of a game management system into an otherwise conventional farming system resulted in a greater increase in numbers of species of conservation concern than numbers of other species. However, game management may also benefit some of these less threatened species, as indicated by Stoate e*t al.* (2000) for Corn Bunting *Miliaria calandra* at the Portuguese study site . Although the English study found little change in species diversity at the farm scale, modifications to farming systems, such as the integration of game management, can therefore contribute to maintaining species diversity at the national scale.

Extensive traditional farming systems are currently receiving support in order to meet social and environmental objectives under EU Rural Development Regulation 1257/99. Borralho *et al*. (1999) indicate that a Zonal Programme introduced under an earlier regulation (2078/92) has successfully contributed to bird conservation in the Portuguese study area. In many parts of northern Europe, and in some parts of southern Europe, other innovative approaches to agricultural management may be more acceptable to farmers than a perceived 'reversion' to traditional systems. This is especially likely where management changes have economic, social or cultural benefits to farmers, as well as meeting environmental objectives such as the maintenance of national bird species diversity on farmland.

*Acknowledgements*. The Portuguese study was part of a wider project on biodiversity of European farming systems and was funded by ERENA and the European Commission Environment Programme (PL93-2239). Peter Eden provided agricultural information. The Game Conservancy Trust and the Allerton Research and Educational Trust funded the English study. We thank Nicholas Aebischer for statistical advice and comments on the text.

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**Ornis Hungarica 12-13: 41-50. 2003**

# Bird population changes in Latvian farmland, 1995-2000: responses to different scenarios of rural development

A. Aunins and J. Priednieks

Aunins, A. and Priednieks, J. 2003. Bird population changes in Latvian farmland, 1995-2000: responses to different scenarios of rural development. – Ornis Hung. 12-13: 41-50.

After the collapse of the collective farm-based agricultural production system in Latvia during the early 90s, the agricultural sector reached its lowest point in the mid-90s. After 1995, some regions were showing various signs of agricultural recovery while others were experiencing further abandonment. A point count-based system for monitoring bird populations in an agricultural landscape was established in 4 geographically, structurally and economically different regions of Latvia in 1995, as was a scheme for mapping land use changes. Each of the 4 study areas has followed a different scenario of rural development during the study period. Our study analyses the changes of the species' populations and land use during the last 6 years revealing patterns common to all areas as well as prominent differences between them. Populations of several bird species changed considerably during the study period, as did the composition and area of most habitats. There was a general tendency for arable lands to increase whereas grasslands (especially meadows) and cattle enclosures decreased. The increase in abandoned land area peaked in 1997 but stabilised or started to decrease afterwards. However, the initial habitat distribution and the degree of the above changes varied between the areas, thus differently affecting bird populations within the study plots. The diverse patterns and sources of development and of bush clearance made these differences even more prominent.

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## 1. Introduction

Populations of many farmland birds have declined dramatically in Western Europe (Flade & Steiof 1990, Saris *et al.* 1994, Fuller *et al.* 1995). Numerous papers have analysed these processes and have found that most of the factors are related to intensification of agriculture (*e.g.* Chamberlain *et al.* 2000, Donald *et al.* 2001). It has been acknowledged that cereal yields alone explained over 30% of variation in bird population trends (Donald *et al.* 2001) and thus can be used as a measure of agricultural intensity in arable lands.

The processes in the agricultural sector developed differently in Eastern Europe. The intensity of Latvian agriculture has never been as great as in many EU countries, where cereal yields exceeded 60 quintals per hectare (q/ha) (FAOSTAT Database). After the collapse of the moderately intensive collective farm-based system in the beginning of the 1990s, agricultural production in Latvia reached its lowest point in 1995 (Anon 1996a).

Cereal yields decreased from 23.3 to 16.6 q/ha, cattle numbers decreased by 70% and usage of mineral fertilisers and pesticides decreased by almost 90% at that time (Anon 1996a, 1999a). A more detailed overview of agriculture in Latvia is given in Aunins *et al.* (2001).

Unfortunately monitoring data on bird populations in agricultural lands are scarce for the period 1990-1995 (Priednieks, unpublished data) when these dramatic changes occurred. Thus the recovery and rapid increase of many bird species like Grey Partridge *Perdix perdix*, Kestrel *Falco tinnunculus* and others associated with farmland remain undocumented.

The principal purpose of the present study was to analyse changes of bird populations in Latvian farmland and the possible factors causing these changes. The species-habitats relationships and the importance of different habitats or landscape features have been reported earlier (Priednieks *et al.* 1999, Aunins *et al.* 2001).

## 2. Study Area and Methods

### Field studies

The field studies were conducted in 1995-2000 in four areas (Fig. 1). All study areas are located in mixed farmland, each having a size of 100 km$^2$. They are located in different regions of Latvia, have different landscape structure and were selected to be representative of the dominant farming practice in each region. Together they create a gradient of farming intensity that is representative of Latvian farmland as a whole. The two


Fig. 1. Location of the study areas.

westernmost study areas are located in regions of intensive farming but each has different landscape structure. Jelgava has very low percentage of forests and shrubland, most of its territory being used for agriculture. Blidene has a very mosaic landscape structure that is comprised of a large percentage of forests and shrubland, the presence of wetlands being characteristic. The other two areas are in areas of low intensity agriculture. The northern area (Skulte) has large percentage of woodlands and shrubland. Most former arable land is abandoned. The eastern area (Teichi) has lower percentage of forests and shrubland, and has still maintained a large percentage of natural (including floodplain) meadows. A more detailed comparison of the study areas has been given in Priednieks *et al.* (1999).

In each of the four study areas, bird count points were chosen randomly using a grid pattern layout as described in Priednieks *et al.* (1999) & Aunins *et al.* (2001). Only the 160 points (40 in each area) that were counted all six study years were included in the analyses.

At each census point, five-minute bird

Tab. 1. Relative occurrence of habitats and landscape features within the described 200 m zones around bird count points of the study areas (mean measurements over the six years taken).

| | Blidene | Jelgava | Skulte | Teichi |
|---|---|---|---|---|
| *Habitats measured and displayed as % of area* | | | | |
| Winter cereals | 15.4 | 21.6 | 3.6 | 9.0 |
| Summer cereals | 5.5 | 23.7 | 13.0 | 17.2 |
| Root crops | 3.5 | 9.6 | 4.3 | 3.2 |
| 1st year fallow | 2.5 | 3.8 | 2.6 | 3.8 |
| Abandoned lands | 19.9 | 7.6 | 21.4 | 13.4 |
| Sown grasslands | 10.3 | 15.9 | 22.8 | 22.6 |
| Improved meadows and pastures | 12.6 | 3.8 | 9.3 | 1.9 |
| Dry and moderately moist natural meadows | 11.6 | 1.1 | 2.8 | 11.7 |
| Wet natural meadows | 3.2 | 3.1 | 1.8 | 1.1 |
| Ponds and pools with emergent vegetation. | 2.4 | 0.1 | 0.2 | 1.1 |
| Ponds and pools w/o emergent vegetation | 0.2 | 0.1 | 0.0 | 0.5 |
| Forests | 4.6 | 2.9 | 9.8 | 6.7 |
| Orchards | 0.7 | 1.0 | 0.3 | 0.0 |
| Shrubs | 6.3 | 0.1 | 2.8 | 1.8 |
| Farmsteads | 1.1 | 2.9 | 4.3 | 3.4 |
| Isolated farm buildings | 0.0 | 1.7 | 0.4 | 1.0 |
| Ruderal areas | 0.1 | 0.9 | 0.8 | 1.4 |
| *Habitats measured as length (m), displayed as density (m/ha)* | | | | |
| Clean ditches | 7.8 | 11.5 | 6.7 | 17.7 |
| Ditches with bushes | 8.5 | 14.8 | 10.5 | 8.7 |
| Natural rivers | 1.1 | 3.2 | 1.5 | 0.3 |
| Alleys | 0.5 | 2.3 | 5.1 | 0.3 |
| Linear shrub belts | 3.5 | 0.9 | 6.6 | 4.7 |
| Roads | 25.9 | 28.3 | 28.6 | 32.6 |
| Electric and telephone lines | 12.3 | 23.9 | 41.1 | 33.7 |
| Enclosures and fences | 0.0 | 1.4 | 1.7 | 19.2 |
| *Features counted as absolute numbers, displayed as number per 100 ha* | | | | |
| Small ponds and pools with emergent vegtn | 0.8 | 0.3 | 0.2 | 2.9 |
| Small ponds and pools w/o emergent vegtn | 0.4 | 0.2 | 0.3 | 0.7 |
| Separate trees | 25.2 | 6.7 | 9.8 | 18.1 |
| Separate bushes | 17.6 | 16.3 | 15.0 | 34.1 |
| Stone and brushwood heaps | 1.3 | 0.2 | 1.3 | 8.3 |

counts (no limitation was placed on the horizontal distance at which birds were reported) were performed twice per season, at around mid-May and mid-June, respectively. Migrants and other birds flying high above the site were excluded from further analysis.

The total number of species recorded per point was used as a measure of species richness. For each point and species, the number of birds recorded was interpreted in pairs (*e.g.* Two singing birds were considered as two pairs, whereas one bird singing and one bird observed (if not an obvious male) were considered as one pair). The higher of the two counts obtained was used.

The area within a circle of radius 200 m (area 12.56 ha) around each point was described by means of 30 habitat variables. The variables, their units of measurement, and their relative abundance within the described zones are shown in Tab. 1. Because the count points were distributed only in agricultural land, the proportions of habitats within the described 200 m zones differ from general landscape characteristics given above.

Tab. 2. Changes in land use and occurrence of landscape features in the four study areas (1995-2000).

| | Blidene | Jelgava | Skulte | Teichi |
|---|---|---|---|---|
| *Habitats measured as % of area* | | | | |
| Winter cereals | ++ | +++ | +++ | +++ |
| Summer cereals | ---(F) | + | 0(F) | -- |
| Root crops | 0 | ++ | ++(F) | -- |
| 1st year fallow | F | F | F | F |
| Abandoned lands | +(F) | +++(F) | ++(F) | +++ |
| Sown grasslands | +(F) | --- | - | ++ |
| Improved meadows and pastures | --- | --- | --- | ++ |
| Dry and moderately moist natural meadows | 0 | -- | -- | --- |
| Wet natural meadows | - | 0 | ++ | - |
| Ponds and pools with emergent vegetation | - | 0 | 0 | - |
| Ponds and pools without emergent vegetation | ++ | 0 | 0 | ++ |
| Forest | 0 | 0 | 0 | - |
| Shrubberies | -- | 0 | + | ++ |
| *Linear habitats* | | | | |
| Clean ditches | 0 | --- | --- | - |
| Ditches with bushes | - | +++ | ++ | ++ |
| Natural rivers | --- | 0 | 0 | 0 |
| Linear shrub belts | - | 0 | 0 | -- |
| Alleys | 0 | --- | 0 | +++ |
| Roads | 0 | 0 | 0 | 0 |
| Enclosures and fences | 0 | --- | - | - |
| Electric and telephone lines | + | - | 0 | -- |
| *Point objects* | | | | |
| Separate trees | -- | 0 | + | 0 |
| Separate bushes | --- | + | 0 | + |
| Stone and brushwood heaps | +++ | 0 | - | - |
| *Habitat groups* | | | | |
| Active arable | ++ | ++ | +++ | - |
| Active arable incl. sown grass | ++ | 0 | ++ | 0 |
| Meadows | -- | --- | --- | -- |
| Meadows and abandoned | - | - | -- | 0 |

0 = change does not exceed 5%
+ or - = change between 5 and 20%
++ or -- = change between 20 and 50%
+++ or --- = change exceed 50%
F = fluctuating

We used the periodicals of the Central Statistical Bureau of Latvia (Anon 1996b, 1997, 1998, 1999b, 2000) as an information source on annual yields in the relevant districts (1995-1999), but these figures should be treated with care because they are not representative of all types of farming, being biased towards state farms and statutory companies. Nevertheless, they represent the regional differences quite well.

**Statistics**

TRIM version 3 software (Pannekoek & van Strien 2001) was used for analysis of bird count data. The following models were tested for each species (with 1995 as the reference year): no time effect (N), linear trend without covariates (L), linear trend including the study area as covariate (LC), linear trend without covariates and with stepwise selection of changepoints

(LT), and linear trend including the study area as covariate and stepwise selection of changepoints (LTC). Level $P \leq 0.05$ was used as significance criterion in Wald tests to enter or remove the changepoints in the stepwise procedures. Models that included the study area as a covariate were rejected if the value of the Wald test for significance of covariate exceeded $P=0.20$. The remaining models were compared and the model that gave the best fit according to Likelihood Ratio was chosen. In the few cases when several models gave maximum fit according to this test ($P=1.000$), the model with the smallest Akaike's Information Criterion was chosen. The modelled indices were used for estimating population status.

An attempt to use the TRIM software for analysing habitat changes was made, but almost all models were rejected, significance being $P<0.001$.

## 3. Results

### Changes in habitats and farming intensity

All the study areas experienced significant changes in land use and the abundance of several landscape features during the six study years (Tab. 2). A steep decrease in meadows was common to all areas, being caused both by abandonment and conversion to arable land. However, there were different patterns of change in the 3 categories of meadows. Blidene did not experience significant decreases of dry and moderately moist natural meadows. Although conversion to arable land persisted, it was balanced by the introduction of mowing, grazing in previously abandoned lands, or both. The main meadow losses in this area were experienced in the category of improved meadows and pastures. Conversion of meadows to arable land was most severe in Jelgava & Skulte, but was less so in Teichi where the decrease in dry and moderately moist natural meadows was caused mainly by their natural improvement and encroachment by bushes after abandonment. An increase of abandoned land was common to all areas to various extents. However, note that the main increase occurred between 1995 and 1997, after which period the rate of abandonment stabilized or started to decrease, except in Teichi where it increased.

An increase in winter cereals was observed in all areas. Only Jelgava experienced increases of other crop types that fluctuated or decreased in the other areas. However, the area of active arable lands increased in all three western study areas.

An important source of differences between the study areas was reflected by changes in distribution of various shrub-dominated habitats (shrubland, ditches with bushes, linear shrub belts and isolated bushes). All these habitats decreased in Blidene and either remained stable or increased in Jelgava or Skulte. The main source of increase was ditches becoming overgrown. In Teichi bush encroachment took place in meadows, abandoned lands and ditches. At the same time, roadside shrub belts decreased. Jelgava experienced cutting down of roadside tree lines (alleys) whereas in Teichi new alleys appeared after removing the roadside bushes and not removing the trees. All study areas experienced reductions in cattle enclosures and other fences as a result of the continuous decrease in livestock keeping.

Tab. 3. Mean number of bird species registered per point and total number of species registered in the study areas.

| Study area | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | Mean |
|---|---|---|---|---|---|---|---|
| Mean number of bird species registered per point | | | | | | | |
| Blidene | 15.20 | 13.45 | 13.78 | 13.68 | 14.55 | 15.55 | 14.37 |
| Jelgava | 11.25 | 12.60 | 11.58 | 12.28 | 11.35 | 11.45 | 11.75 |
| Skulte | 14.78 | 16.25 | 14.78 | 16.70 | 14.25 | 15.93 | 15.45 |
| Teichi | 14.41 | 16.61 | 17.71 | 17.29 | 20.46 | 21.15 | 17.94 |
| Total | 13.91 | 14.74 | 14.48 | 15.00 | 15.19 | 16.05 | 14.90 |
| Mean number of bird species registered per study area | | | | | | | Total |
| Blidene | 77 | 65 | 69 | 67 | 70 | 71 | 104 |
| Jelgava | 63 | 62 | 65 | 59 | 57 | 57 | 85 |
| Skulte | 68 | 60 | 62 | 62 | 61 | 65 | 97 |
| Teichi | 73 | 76 | 69 | 72 | 70 | 72 | 101 |
| Total | 105 | 96 | 95 | 96 | 94 | 102 | 134 |

The intensity of farming (measured by yields) varied between the study areas as well as changing during the study period. The highest winter cereal yields were found in Blidene & Jelgava (31.5 and 30.5 q/ha on average), the values reflecting increasing yields (by 1.6 and 2.8 q/ha respectively). Winter cereal yields in Skulte & Teichi were much lower (19.3 and 15.7 q/ha respectively), the yield in Teichi decreasing significantly by 5.1 q/ha). A rapid growth of yields in Skulte was recorded between 1995 and 1997, followed by a decline, after which the 1999 yields approximated the 1995 levels (an increase of 0.2 q/ha). Summer cereal yields fluctuated synchronously in all study areas without any pronounced tendency, but they were higher in Blidene & Jelgava (23.0 and 23.3 q/ha on average) compared to Skulte & Teichi (13.9 and 12.1 q/ha). Yields of grass production also were higher in Blidene & Jelgava (45.0 and 39.8 q/ha) than in Skulte & Teichi (32.6 and 30.6 q/ha). Although the year-by-year numbers fluctuated, there was a tendency for the grass production yields to grow in Blidene & Skulte and to decline in Jelgava & Teichi.

## Changes in bird populations

The mean number of species registered per point was stable in all study areas except Teichi (Tab. 3) where it increased from 14.4 in 1995 to 21.2 in 2000. At the same time the total number of species registered per study area did not increase in any of the study areas (but slightly decreased in Jelgava).

The analysis of the bird population changes is summarized in Tab. 4. Some species (*e.g.* Quail *Coturnix coturnix*, White Wagtail *Motacilla alba*, Sedge Warbler *Acrocephalus schoenobaenus*, Thrush Nightingale *Luscinia luscinia*) show a common change pattern in all study areas suggesting that populations of these species currently are more affected by large-scale factors than by area-specific factors. However, population change patterns for most of the species differ between the study areas suggesting that area-specific factors play important roles there.

In general, increases of shrub and forest generalist species are obvious and differences between the study areas are not as pronounced as for other groups. These

Tab. 4. Trends of bird populations in study areas (1995-2000).

| Species | Registr ations | Blidene | Jelgava | Skulte | Teichi | Total | Best model |
|---|---|---|---|---|---|---|---|
| Open agricultural land (arable, grasslands, abandoned lands) | | | | | | | |
| White Stork *Ciconia ciconia* | 494 | -- | -- | ++ | +++ | + | LC*** |
| Quail *Coturnix coturnix*[1] | 36 | +++ | +++ | +++ | +++ | +++(F) | LT(1)*** |
| Corncrake *Crex crex* | 310 | -- | + | ++(F) | +++ | ++(F) | LCT(4)$^{ns}$ |
| Lapwing *Vanellus vanellus* | 505 | F | F | +++ | F | +? | N/A |
| Skylark *Alauda arvensis* | 5245 | -- | + | ++ | ++ | + | LC*** |
| Meadow Pipit *Anthus pratensis* | 681 | ++ | - | -- | -- | -- | LC** |
| Whinchat *Saxicola rubetra* | 877 | ++ | + | ++ | 0 | + | LCT(1)* |
| Shrubby edge of agricultural land | | | | | | | |
| Grasshopper Warbler *Locustella naevia* | 149 | F | F | F | F | +++(F) | LT(2)** |
| Red-backed Shrike *Lanius collurio* | 124 | + | + | +(F) | + | +(F) | LT(3)*** |
| Scarlet Rosefinch *Carpodacus erythrinus* | 485 | -- | -- | -- | +++ | - | LCT(1)* |
| Yellowhammer *Emberiza citrinella* | 983 | - | -- | --- | +++ | - | LCT(2)*** |
| Species feeding on agricultural lands | | | | | | | |
| Buzzard *Buteo buteo* | 239 | - | --- | --- | +++(F) | 0(F) | LCT(2)** |
| Woodpigeon *Columba palumbus* | 259 | 0 | 0(F) | 0(F) | +++ | ++ | LCT(1)$^{ns}$ |
| Fieldfare *Turdus pilaris* | 155 | 0 | 0 | 0 | 0 | 0 | LT(2)$^{ns}$ |
| Farmsteads | | | | | | | |
| White Wagtail *Motacilla alba* | 222 | --- | --- | --- | --- | --- | LT(2)*** |
| Icterine Warbler *Hippolais icterina* | 115 | +++(F) | --(F) | ---(F) | +++(F) | +(F) | LCT(1)*** |
| Starling *Sturnus vulgaris* | 777 | -- | ++(F) | +++(F) | --(F) | + | N/A |
| Goldfinch *Carduelis carduelis* | 143 | 0 | 0 | 0 | 0 | 0 | N* |
| Linnet *Accanthis cannabina* | 112 | +++ | -- | --- | --- | -- | LC* |
| Wetlands | | | | | | | |
| Marsh Harrier *Circus aeruginosus* | 66 | 0 | 0 | 0 | 0 | 0 | N** |
| River Warbler *Locustella fluviatilis* | 143 | 0 | 0 | 0 | 0 | 0 | N** |
| Sedge Warbler *Acrocephalus schoeniclus* | 168 | -- | -- | -- | -- | -- | L* |
| Reed Bunting *Emberiza schoeniclus* | 146 | 0 | 0 | 0 | 0 | 0 | N** |
| Shrubberies | | | | | | | |
| Thrush Nightingale *Luscinia luscinia* | 979 | +++ | +++ | +++ | +++ | +++ | LT(2)** |
| Marsh Warbler *Acrocephalus palustris* | 747 | +++ | --(F) | +++(F) | +++(F) | ++(F) | LCT(5)*** |
| Whitethroat *Sylvia communis* | 1162 | +++ | ++ | ++ | +++ | +++ | LC*** |
| Garden Warbler *Sylvia borin* | 367 | +++ | +++ | +++ | + | +++ | LTC(3)*** |
| Forest | | | | | | | |
| Cuckoo *Cuculus canorus* | 505 | +++ | +++ | +++ | +++ | +++ | LT(3)*** |
| Tree Pipit *Anthus trivialis* | 502 | -- | +++ | +++ | +++ | +++ | LC*** |
| Blackcap *Sylvia atricapilla* | 97 | ++(F) | ++(F) | ++(F) | ++(F) | ++(F) | L** |
| Willow Warbler *Phylloscopus trochilus* | 304 | -- | -- | -- | -- | -- | LT(3)*** |
| Chiffchaff *Phylloscopus collybita* | 127 | +++(F) | +++ | +++ | +++ | +++ | LT(1)*** |
| Blackbird *Turdus merula* | 463 | - | ++(F) | --- | +(F) | -(F) | LCT(4)*** |
| Song Thrush *Turdus philomelos* | 371 | -- | - | +++ | +++ | +++ | LCT(3)*** |
| Redwing *Turdus iliacus* | 103 | +++ | +++ | +++ | +++ | +++ | LT(1)* |
| Golden Oriole *Oriolus oriolus* | 499 | ++ | +++(F) | +++ | +++ | +++ | LCT(3)*** |
| Great Tit *Parus major* | 144 | +++ | ++ | 0 | +++ | +++ | LCT(3)*** |
| Chaffinch *Fringilla coelebs* | 957 | +++ | - | +++ | +++ | +++ | LCT(2)*** |
| Declining | | 13 | 13 | 10 | 6 | 8 | |
| Increasing | | 16 | 16 | 20 | 24 | 24 | |

[1] Population of the species was stable at a very low level 1995-1999
N = no time effects
L = linear trend
LC = linear trend, significant differences between study areas
LT = linear trend with significant changepoints, number of changepoints are given in brackets
LTC = linear trend with significant changepoints, significant differences between study areas, number of changepoints are given in brackets
N/A = all models rejected with significance P<0.05, expert judgement used for estimation of trends
0 = stable (change does not exceed 5%)
+ or - = slight increase or decline (change between 5 and 20%)
++ or -- = moderate increase or decline (change between 20 and 50%)
+++ or --- = strong increase or decline (change exceed 50%)
F = fluctuating
*, **, *** = model goodness-of-fit (significance of likelihood ratio test - P>0.95, P>0.99, P>0.999 accordingly)

increases can be associated with the general increase of forest and shrub areas in Latvia due to encroachment of abandoned lands. No such increase can be observed in species groups of agricultural and wetland habitats where the proportion of species having declining trends is larger and differences between the study areas are more pronounced.

Jelgava & Blidene have larger numbers of declining species than the other two areas (Tab. 4). Teichi had the smallest number of such species, half of which were those declining in all areas. This area also had the largest number of increasing species, the difference being due mainly to species of agricultural habitats.

## 4. Discussion

A six-year period is too short a time span to indicate clear trends that would describe current tendencies for the farmland bird populations for the whole of Latvia. A large proportion of the changes are caused by yearly fluctuations in numbers due to the influence of various abiotic and biotic factors such as weather conditions (both in wintering areas and breeding grounds), availability of a variety of resources, and nesting success in the previous breeding season (Wiens 1989, Fuller 1994). This conclusion mostly applies to species whose best models do not include the study area as a significant covariate (Tab. 4). However, the large proportion of species whose changing patterns differ significantly between the study areas suggests that local processes play very important roles. These changes in breeding bird populations during the study period chiefly have been caused by changes in distribution of agricultural habitats and various landscape features and by changes in farming intensity. In this respect, all the study areas have undergone different scenarios of development.

The only area that experienced decreases not only of the area of active arable lands (Tab. 2), but also of farming intensity, was Teichi. However, the decrease of arable lands was balanced by increase of sown grasslands, and the decrease of meadows by the increase in abandoned lands. Thus the proportion of cultivated and uncultivated areas remained approximately the same. As the total number of species did not increase we believe that the increase of the mean number of species registered per point in this study area occurred due to the increase of shrub-dominated habitats and the decrease of farming intensity. Although encroachment by bushes took place both in ditches and abandoned lands, it did not affect negatively open habitat species, yet here the increase in abandoned lands was more pronounced (Tabs 2 and 4). However, if this area continues to develop this way, it inevitably will lead to a reduction of total open area and a decline of open habitat species.

The other area with low farming intensity (Skulte) has experienced an increase of arable land (*cf* winter cereals) and a strong decrease of grassland areas. The increase in farming intensity has been insignificant and shrub encroachment has been recorded both for abandoned fields and ditches. Unlike Teichi, this area did not experience any rapid increase in the number of species registered per point. Rather, decreases were observed of several typical agricultural species that were increasing in Teichi.

The two westernmost areas are similar to each other; both are more intensively farmed than others and experienced further intensification during the study period, as expressed by increases of yields and of the area of arable land. However, the areas differ very much in their landscape structures, proportions of farmland habitats and the change pattern of shrub-dominated habitats. Nevertheless, in both areas more than twice as many species are decreasing than in Teichi, most of them being associated with agricultural habitats. Although farming intensity is not even close to that in EU countries yet, we expect many private farmers will start, or have started, to use western farming practices that have been a principal cause of declines of most farmland bird species populations in western Europe. Our results, however, are based on the state statistics that are biased towards state and statutory farms, and therefore cannot show the full picture. Although all shrub-dominated habitats decreased in Blidene, it is interesting to note that the species associated with them continue to increase. We explain this paradox as a result of the still-continuing expansion of these habitats in Latvia as a whole, due to widespread encroachment of former arable lands, thus providing these species with ideal living niches, increasing their reproductive success to allow overproduction to export surplus birds to neighbouring sub-ideal habitats.

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# Bird community dynamics in a primaeval forest - is interspecific competition important?

T. Wesołowski

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Views on the role of interspecific competition in shaping the structure and dynamics of bird communities vary widely, from negligible impacts on one hand, to being the leading ecological and evolutionary force, producing highly structured communities on the other. What role has this factor played in forming pristine European forest bird communities? Data collected in the primaeval temperate forest, the Białowieża National Park (E Poland) over a period of 25 years are used to answer this question. The bird community of the Białowieża Forest was composed of numerous species, usually breeding at low densities. Food resources and nest sites were usually superabundant, but production of young remained low, due to heavy nest predation. The population sizes of individual species/guilds changed either independently of each other or in parallel. These results indicate that interspecific competition has apparently been of minor importance in the primaeval conditions. This remains in sharp contrast to its frequently dominant role observed in secondary woods that contain nest boxes. The implications of these findings for our understanding of biological processes are discussed.

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## 1. Introduction

Deciduous and deciduous-coniferous forests of the temperate zone have undergone extermination, fragmentation or deep transformation well before the origin of ornithological research. Our knowledge of forest bird biology in pristine conditions is therefore full of gaps or misinterpretations. Consequently, research conducted in remnants of the ancient lowland temperate forests is of utmost importance and provides the reference points or baseline against which comparisons can be made with data collected in habitats transformed by human activity. This principle lay behind the decision to launch in 1975 a programme of ornithological studies in the Białowieża Forest, in which the last fragments of European primaeval temperate lowland forest are to be found. It was hoped that the old-growth primaeval forest stands preserved in the strictly protected part of the Białowieża National Park (BNP) would constitute 'a window into the past' through which one would gain insights into the ecology of pristine temperate plant and animal communities. The aims of the studies were to describe patterns found in primaeval forest breeding bird communities and to understand the ecological and behavioural processes generating the patterns.

To achieve these broadly defined goals, data on breeding bird densities were gathered in permanent study plots distributed in all types of old-growth stands in the

BNP. Censuses, using a modified 'combined mapping' technique (Tomiałojć 1980a; a method of producing near-absolute estimates), were repeated every year from 1975, producing a 25+ year data series. In addition to data on birds, information was gathered also on the variation in those environmental variables that might be relevant, such as weather, leaf-eating caterpillars, tree seed crops, holes and small mammals. The results of the first 20 years of census work have been summarised in a series of papers (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1990, 1994 and 1996; Wesołowski & Tomiałojć 1997). In addition to this long-term community-wide approach, population studies of 12 individual species were carried out for shorter periods (see list of References); their results allowed one to gain better insights into processes that could have generated the observed patterns.

In the mid-1970s when the project began, the unitarian 'competitive' view of bird communities (as equilibrial, stable, strongly interacting units), as marshalled by MacArthur (1972) and his followers, prevailed. The opposing 'individualistic' model (communities non-equilibrial, comprised of loosely-knitted sets of species, changing numbers independently of one another, individuals responding to a varying array of factors) was rather uncommon (reviewed in Wiens 1989). Current views of animal community structure are much more pluralistic (McIntosh 1995), recognising that different processes can interact in shaping the structure and dynamics of communities. The pure 'equilibrial' and 'individualistic' models are now treated as special cases, lying at opposite ends of the continuum.

Results from bird studies in European woodlands transformed by human activity demonstrate that interspecific competition, both exploitation and interference types, can be common there. Interspecific competition is especially important in shaping numerical relationships among hole-nesting birds (reviewed in Newton 1998). Does this picture hold also true under the conditions prevailing in the BNP? How important has been proximately acting interspecific competition in forming breeding bird communities in pristine European forest? I shall come to these questions after introducing the Białowieża Forest and its breeding avifauna.

## 2. Białowieża Forest

The Forest is situated on the border between Poland and Belarus. Of its total area of 1250 km$^2$, some 580 km$^2$ belong to Poland. The geographic co-ordinates of Białowieża village (52°41'N, 23°41'E) correspond to the latitudes of Berlin and London. The climate is subcontinental, producing long snowy winters, the snow cover lasting usually for about three months, although almost snowless winters occasionally do occur. Biogeographically the Forest falls within the mixed forest (deciduous-coniferous) zone that contains a significant amount of native Norway spruce *Picea abies* in almost all types of tree-stands. For more detailed description, see Faliński (1968, 1986), Tomiałojć (1991), Tomiałojć *et al*. (1984), Tomiałojć & Wesołowski (1990, 1994) and Jędrzejewska & Jędrzejewski (1998).

The Białowieża Forest constitutes a remnant of the vast European lowland forests that once extended across the continent. Probably being the least changed remnant, it contains the largest amount of pristine features of any forest complex existing in temperate Europe, its relatively good state of preservation stemming from a long chain of fortunate historical events. Though people have inhabited the Białowieża Forest region since Neolithic times, colonisation of the forest complex proper became more intensive from only the 10$^{th}$ century onwards (Faliński 1968). Large-scale timber extraction in the Forest did not begin until the 1914-18 World War. Nevertheless, even by 1921, a 47.5 km$^2$ patch of the most diversified and best-preserved stands had been excluded from forestry use and declared a strictly protected nature reserve (currently within the BNP; see Tomiałojć & Wesołowski (1994) for details).

The old-growth stands preserved in the BNP are distinguished from those in other temperate forests by these features:

1. They are multi-storied, mixed-species and unevenly aged (the oldest trees dating from 1500-1600).
2. They contain trees reaching unusual heights (the tallest spruces reaching 57 m and several other species 42-45 m).
3. They contain a large amount of undisturbed dead timber and uprooted trees (the latter structures being very important as nesting substrates (review in Wesołowski & Tomiałojć 1995).

The study plots were situated in three main types of BNP forest habitats, upland deciduous woods of the oak-lime-hornbeam *Tilio-Carpinetum* type (44% of the BNP area) and swampy deciduous (22%) and coniferous stands (28%). Detailed descriptions of the habitats and plots studied in the BNP are given in Tomiałojć *et al.* (1984), Tomiałojć & Wesołowski (1990, 1994, 1996) and Wesołowski & Tomiałojć (1995).

## 3. Breeding bird community of the primaeval forest

So far, 111 forest and forest-edge species have been recorded breeding in the Białowieża Forest, 90 of them (81%) within its strictly protected part (47.5 km$^2$) (Tomiałojć & Wesołowski 1990, T. Wesołowski unpubl). Non-passerines, amongst which are eight raptor species, four owls, eight woodpeckers, Black Stork *Ciconia nigra*, Hazel Grouse *Bonasa bonasia* and Green Sandpiper *Tringa ochropus* form as much as 40% of the BNP avifauna. These values appear to be among the highest recorded for a single forest complex, especially when set against those pertaining to the equivalent western European forests (Tomiałojć & Wesołowski 1990).

Within the permanent census plots (total area 1.9 km$^2$) 84 breeding species have been recorded, 64 of them as annual breeders, and 20 additional and scarce species that have bred at least once since the mid-1970s. With minor exceptions, the species composition has not changed during the whole period (Tomiałojć *et al.* 1984, Tomiałojć 1995, T. Wesołowski unpubl). The number of species breeding within a single plot (sizes 25-33 ha) depended on habitat type and plot position. The highest number - total number of species was 72 (48-52 species/year) - was recorded in the riverine forest at the forest edge, followed by the oak-hornbeam plot at the edge - total number of species was 63 (40 species/year). In the first years of the study, deep in the forest interior, riverine habitat harboured more species (35) than did either oak-hornbeam (29-31) or coniferous habitats (26 species/year) (Tomiałojć *et al.* 1984). However, due to increasing species richness in coniferous habitat in the 1990s these differences ceased to exist, and since then all areas have held 33-36 breeding species annually (Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl).

As a rule, individual species bred in the BNP in low densities, only two species, Chaffinch *Fringilla coelebs* and Robin *Erithacus rubecula* exceeding a density of 5 pairs/10 ha regularly. Examples of species that achieved that level of abundance irregularly are Wood Warbler *Phylloscopus sibilatrix*, Collared Flycatcher *Ficedula albicollis*, Song Thrush *Turdus philomelos*, Hawfinch *Coccothraustes coccothraustes*, Blackcap *Sylvia atricapilla* and Starling *Sturnus vulgaris*, but not necessarily in all habitats. The remaining species occurred in much lower numbers, the mean density range being only 1.4-2.6 pairs/10 ha (Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl). Overall, the breeding densities of the above species in the BNP (see below) are much lower than those recorded in woods transformed by human activity, as they are also for several other species. Densities of Wren *Troglodytes troglodytes* and Dunnock *Prunella modularis* are *c*8 times lower, of Great Tits *Parus major* 10 times, Blackbirds *Turdus merula* 40 times and of Woodpigeon *Columba palumbus* as much as 400 times lower than the highest densities reported from secondary woods (Tomiałojć *et al.* 1977, 1984, Tomiałojć 1980b, Wesołowski 1983, Wesołowski *et al.* 1987).

The overall density of the bird community in the BNP was rather low, reaching only 100-110 pairs/10 ha in the most densely populated areas, 124 pairs/10 ha being the maximum value recorded in a single season (Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl). Densities declined from the forest edge to the interior and from deciduous to coniferous habitats. Densities in the latter, 35-50 pairs/10 ha, were always lowest (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl). In contrast to the late 1970s (Tomiałojć *et al.* 1984), bird community densities increased during the study period, the mean densities in all plots being higher by 13-38% by the late 1990s. These increases resulted from the simultaneous numerical growth of several species, such as Chaffinch, Collared Flycatcher, Blackbird and Marsh Tit *Parus palustris*. The tendency towards parallel changes in species that differ so much in their food requirements, nest sites, and migratory patterns suggests that a number of different causal factors had to be involved, as it would be difficult to conceive of a common denominator, a single factor which could account for all those increases. Neither reduced severity of winters, nor increased seed crops, rodent numbers, or caterpillar abundance (see below) showed long-term trends in the BNP, and so these could be ruled out as causes of the observed increases of bird numbers in the Forest (Wesołowski & Tomiałojć 1997, Wesołowski unpubl).

Irrespective of these temporal changes, in all habitat types, insectivores collecting invertebrates in tree crowns constitute *c*50% of the whole breeding community and ground insectivores (*c*30%) are the second most numerous group. In terms of nesting requirements, approximately half of bird species build open nests in tree crowns, one-third use holes, and the rest breed on the ground or just above it. Due to rather severe winters, short-distance (*c*50%) and tropical migrants (*c*25%) migrants form the bulk of the community (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl).

In summary, the BNP breeding bird community is composed of numerous species, but they usually breed in low to moderate densities. For some reason their numbers tend to change in parallel. Could interspecific competition account for these patterns? Are low densities caused by the proximate interspecific competition for limiting resources, such as the breeding season food supply or nest sites? These questions are explored below.

## 4. Food limitation and competition for food in the breeding season?

Birds collecting invertebrates from leaves and twigs form about 50% of bird assemblages in the BNP and leaf-eating caterpillars constitute a substantial part of their diet. Preliminary observations revealed that these caterpillars constituted 70% of Middle Spotted Woodpecker *Dendrocopos medius* nestling food (in 1978), 40-60% of food brought to Wood Warbler nestlings (in 1987-1988) (L. Jenni unpubl, R. Cisakowski unpubl), up to 80 % of Marsh Tit nestling food and 55% in the case of Nuthatch *Sitta europaea* (in 1998) (Rowiński & Wesołowski 1999). Tomiałojć (1994) observed that even for Blackbird, in which *Lumbricidae* formed the most important food source, the caterpillars still made up 14-32% of nestling diet in 1986-1989. Therefore, there are grounds to assume that variation in the caterpillar numbers should have a strong influence on the breeding birds. The defoliating *Operophtera brumata* caterpillars occurred usually in low numbers, their outbreaks, causing partial to total defoliation of deciduous trees, separated by 8-11 (Wesołowski & Tomiałojć 1997). However, in the intervening years smaller scale outbreaks of other species, providing alternative food sources, could occur (*e.g.* partial defoliation of *Acer platanoides* by *Ptilophora plumigera* caterpillars; Rowiński & Wesołowski 1999). If, as these observations suggest, apart from the few 'bonanza' years, food resources are relatively scarce for several successive years, we should expect to observe frequent food limitation in the breeding season, and signs of interspecific competition for food. These phenomena could find expression in:

1. Compensatory changes in numbers of would-be competitors.
2. Small clutches.
3. Frequent starvation of young.
4. Lack of resource defence polygyny.

Contrary to expectations, changes in numbers of birds in the BNP over the 25-year study period are mostly independent of variation in *Operophtera brumata* abundance, for numbers of but three of 13 crown insectivores were positively correlated with caterpillar abundance. Moreover, changes in numbers of congeners are most often independent of one another (*e.g.* Great Spotted *Dendrocopos major* and Middle Spotted Woodpeckers, Wood Warbler and Chiffchaff *Phylloscopus collybita*) or positively correlated (*e.g.* Pied *Ficedula hypoleuca* and

Collared Flycatchers, Blue *Parus caeruleus* and Great Tits, Blackbird and Song Thrush) (Wesołowski & Tomiałojć 1997, T. Wesołowski unpubl).

Starvation and strong brood reduction was found to be unimportant, even in the low-caterpillar years. The year-to-year variation in nesting success in the eleven primarily insectivorous species studied so far in the BNP was mostly due to predation (review in Wesołowski & Tomiałojć 1995, Wesołowski 2001, L. Tomiałojć unpubl, D. Czeszczewik unpubl). During a 12-year study of Marsh Tit, only in a single season, when cold and rainy weather arrested caterpillar development and the young were in nests ahead of maximum caterpillar availability, was a sharp increase in nest losses recorded that could be attributed to food shortage. However, even then, the impact of food shortage was only indirect (*e.g.* causing the young to beg louder), because in the main, broods were lost to predation or nest soaking and not to starvation (T. Wesołowski & P. Rowiński unpubl).

The clutch size of Białowieża birds was as large, or larger than the highest values recorded elsewhere (Wesołowski 1983, 1985, 1995, 2000; Piotrowska & Wesołowski 1989, Wesołowski & Stawarczyk 1991, Tomiałojć 1994); polygyny was found to be regular in several species in BNP habitats containing higher breeding densities (Wesołowski 1987). Usually 10-20% of bigamous males were recorded there, but in some years there could be up to 40% of bigamists (some even simultaneous trigamists) as found in a Wood Warbler oak-hornbeam area (plot C, 1978). Because none of the predictions has been confirmed, it seems justified to conclude that neither limitation by food shortage in the breeding season nor interspecific competition for food could be major forces shaping the structure and dynamics of insectivorous birds in this primaeval forest. It is not the equivalent of saying that food plays no role in shaping life of birds breeding there, but it stresses that food shortages alone cannot account for low overall breeding bird densities in the BNP.

## 5. Is competition for breeding holes important?

Secondary hole-nesters serve as a classic example of a group of species limited by shortage of nest sites, or interspecific competition (or both) (von Haartman 1971, Perrins 1979, van Balen *et al.* 1982, review in Newton 1994). The highest densities recorded in the BNP for this group were 30-40 pairs/10 ha (Tomiałojć & Wesołowski 1996, T. Wesołowski unpubl), values that are only just as high as those recorded for single species in nestbox areas, for example 40 pairs/10 ha for Pied Flycatcher (Tiainen *et al.* 1984) or 34 pairs/10 ha for Great Tit (Perrins 1979). Could low numbers in the BNP be due to shortage of holes and competition for this scarce resource? If so, then several phenomena should be visible:

1. Suitable holes are occupied every season.
2. Populations of potentially competing species change numbers in compensatory fashion.
3. Frequent interspecific aggression arises over holes and hole usurpation.
4. Only obligatory hole-nesters use holes.

By and large, the Białowieża results do not support these expectations. Different hole-nesting species changed their numbers in parallel or independently of each other (Wesołowski & Tomiałojć 1997). Furthermore, nest-holes are superabundant. Although, due to enormous technical difficulties (accessibility, safety) it has been impossible to produce data on hole density in the BNP, indirect conservative estimates of hole availability clearly indicate their excess. Collared Flycatcher, the most numerous species of hole-nesters in oak-hornbeam habitat (Tomiałojć & Wesołowski 1990), is the secondary hole-nesting species that breeds latest of all. In a detailed study carried out in 1989, Walankiewicz (1991) found that were at least 28 free holes/10 ha available for flycatcher selection. In other words, an average flycatcher female had at least two potential nest holes to choose from. Because all these flycatcher holes had been available to those species that had begun breeding earlier, early breeding species had at least 3 potential holes from which to select. These values represent minimum estimates because they include only the holes used by the birds in 1989 and it is known that birds are often irregular occupants of the numerous holes available (Wesołowski 2001, unpubl), even if one includes holes used by the facultative hole-nesters. Blackbirds breeding in holes were observed regularly in the BNP, almost 50% of their nests in oak-hornbeam habitat being in holes and semi-holes (Tomiałojć 1993). Moreover, Robin, Dunnock and Wren also used holes regularly.

Interspecific aggressive encounters and hole usurpation of one species by another were recorded in the BNP (Tomiałojć *et al.* 1984) but their frequency was quite low. For example, in a three-year study of Nuthatch based on over 160 broods (Wesołowski & Stawarczyk 1991) no case of hole-usurpation was recorded. Similarly no Collared (n=534) or Pied Flycatcher (n=159) males were killed while prospecting for holes. Hybridisation between Collared and Pied Flycatcher (though at only 0.4%, is nevertheless regular) is probably better understood in terms of competition for mates than as a by-product of competition for holes (Walankiewicz & Mitrus 1997, Czeszczewik & Walankiewicz 1999).

All these observations lead one to conclude that the low hole-nesting densities found in primaeval stands of BNP cannot result from limitation by the shortage of nest sites. Therefore, reports from other areas of the effects of nest hole shortages on limiting bird numbers and of strong interspecific competition for holes seem due mostly to by-products of human-induced habitat transformations, and not due to factors that are relevant in primaeval conditions.

## 6. Other possible mechanisms

As shown in the previous chapters, breeding bird densities in the BNP usually remain below levels set by food or nest site availability. Furthermore, proximately acting interspecific competition is of rather minor importance. Therefore, the Białowieża data, do not offer much support to the idea of equilibrial, saturated, strongly competitively interacting bird communities (Lack 1971, MacArthur 1972). The Białowieża breeding bird community seems to be better described by the 'individualistic' community model (Wiens 1989, McIntosh 1995). If not shortage of resources, then what keeps densities low in the BNP? Several mechanisms can be involved, such as:

1. Undersaturation, or too few birds settling in spring to occupy all the available space.
2. Density limitation by territorial exclusion.
3. Low productivity due to high nest predation.
4. The effects of events outside the Białowieża Forest.

Undersaturation can be common among breeding birds in BNP. It is possible that most of scarce and irregular breeders are permanently unable to fill space, though it is impossible to prove that the unoccupied areas really are suitable for them (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1990). The year-to-year changes in distribution of territories, without any corresponding change in habitat structure, demonstrate this phenomenon much better. For example, densely packed Wood Warbler territories filled plot C in 1978 (21 territories), whereas in 1983 this area was almost empty (3 territories) (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1994). Similarly Great Tit in the low number years (Wesołowski *et al.* 1987) and Nuthatch, even in years of relatively high numbers (Wesołowski & Stawarczyk 1991), were not numerous enough for their territories to fill all the available space.

Saturation of habitats at low densities resulting from birds defending large territories is also clearly demonstrated in the BNP. Even birds as small as Wren, Chiffchaff or Marsh Tit can occupy territories covering 8-10 ha, their territory size in oak-hornbeam habit averaging up to 5 ha (Wesołowski 1983, Piotrowska & Wesołowski 1989, T. Wesołowski unpubl). The large territories in this habitat can to some extent reflect the distribution of necessary requisites, such as fallen logs (Wren) or canopy gaps (Chiffchaff), but the size can also improve males' chance of attracting a mate and, by spreading nests in space, can serve as antipredator device (Wesołowski 1987, Wesołowski *et al.* 1987).

Repopulation of plots after the experimental removal of territorial males (Wren, Great Tit and Wood Warbler) (Wesołowski 1981, Wesołowski *et al.* 1987, Wesołowski & Tomiałojć 1995) shows that if some males can be prevented from establishing territories, limitation of numbers by territorial defence can produce densities as low as 1.5 territories/10 ha.

For the majority of birds studied in the BNP so far (Wesołowski 1983, 1985, 1995, 1998; Piotrowska & Wesołowski 1989, Wesołowski & Stawarczyk 1991, Walankiewicz 1991, Walankiewicz *et al.* 1997, Tomiałojć 1994, Jędrzejewski *et al.* 1994), the overall nest loss rate amounted to 50-70%, occasionally rising to 76% (Wood Warbler) or dropping to 15% (Marsh Tit). For every one of these species, the Białowieża loss rates are equal to or higher than the highest values recorded for this species in other areas more transformed by human activity. Predation is responsible for at least 70-95% of nest losses. The impact of predation is greater if the timing of nest destruction is at a late stage of breeding, as is the case in the BNP, where they are destroyed mostly during the nestling period (Wood Warbler, Chiffchaff, Wren, Nuthatch, Marsh Tit), when the majority of energy investment necessary for rearing a brood has already been input, and when much of the time which otherwise could have been devoted to rearing a replacement brood had been wasted. In other areas, the maximum rate of nest predation does not occur during the nestling period. In consequence of the heavy predation pressure in the BNP, the production of young per breeding pair (and especially the production of young per unit area) is usually very low, sometimes being an order of magnitude lower than recorded in other areas (Wesołowski 1983, Wesołowski *et al.* 1987). The depressed productivity, even without heavy mortality in the intervening non-breeding season, translates into fewer potential recruits each following spring. In turn, this could result in undersaturation and low breeding numbers.

The situation in the BNP as described above has been presented under the assumption that either there are no significant external inputs, or the changes of environmental factors in the Forest reflect their variability over larger areas. However, in spite of the BNP's relatively large area, birds breeding within it do not constitute fixed demographic units. Immigration of birds has contributed, at least in some years, to numerical increases in the highly variable Siskin *Carduelis spinus* and Wood Warbler, but its impact was discernible in some seasons on several other species whose rate of numerical increase in consecutive seasons could not always be accounted for by local production (Wesołowski & Tomiałojć 1997). One might expect emigration to occur in other years as well, though this would be more difficult to demonstrate. If events taking place outside the Forest were acting so strongly that the effects of local factors were swamped, the structure of the breeding community in the BNP could not be explained by analyses carried out solely within the BNP. This explanation seems scarcely to be applicable to Nuthatch (Wesołowski & Stawarczyk 1991) and other permanent Forest residents. But such species constitute only *c*10% of the breeding bird community in the BNP (Tomiałojć & Wesołowski 1996) and for the 90% of the population spending the winter outside the breeding areas, such a conjecture seems more plausible. However, if conditions in the non-breeding period were of the utmost importance for population dynamics, then one would expect species wintering in the same regions to have similar patterns of numerical change. This prediction, though, has not been corroborated (Wesołowski & Tomiałojć 1997). It would be premature to reject this hypothesis altogether, as migratory categories used in the analysis were quite broad, and species included in the same guild could have totally non-overlapping wintering ranges. Nevertheless, there is no evidence at the moment indicating that events in the non-breeding areas are the most influential in controlling bird numbers in the BNP (Wesołowski & Tomiałojć 1997).

## 7. Primaeval versus secondary forests

A comparison of features of bird communities from primaeval BNP stands with those in secondary forests shaped by human activities in other temperate areas of Europe reveals sharp differences between them (Tab. 1), yet both forest types are composed largely of the same tree species and are inhabited by the same bird species (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1990,

Tab. 1. Major differences between primaeval and secondary temperate forests (modified after Tomiałojć *et al.* (1984), Tomiałojć & Wesołowski (1990).

| | **Primaeval** | **Secondary** |
|---|---|---|
| Forest size | large, continuous | fragmented, isolated |
| Predator diversity | high | low |
| Availability of holes | excess | shortage |
| Species richness | high | low |
| Production of young | low | high |
| Density | low | high |
| Interspecific competition | seldom, insignificant | frequent, eminent |

Tomiałojć 2000). The differences stem most probably from anthropogenic causes: in the secondary forests, a combination of fragmentation effects, a simplification of forest structure and widespread predator extermination. On one hand, the anthropogenic factors led to extinctions of more sensitive (large, specialised) species and thus to declining species richness. On the other hand they permitted the more productive populations of surviving species to burgeon, to increase numbers to a level at which limitation by resources and interspecific competition becomes important (Tomiałojć 1980b, 2000; Wesołowski 1983). The high productivity and high bird densities in the secondary woods seem to be characteristic of all habitats in which predator pressure is reduced (islands, colonies, some human settlements) and not characteristic features of temperate forest bird communities. As the Białowieża data clearly demonstrate, in the temperate zone and in pristine conditions, the forest bird communities have exhibited characteristics attributed usually only to tropical forests (high predation pressure, low densities, high species richness). This should be born in mind when one attempts large-scale intercontinental comparisons. For valid comparisons one should compare only the equivalent states. It seems best to avoid drawing conclusions if data from primaeval tropical forest is equated with those gathered in fragmented secondary European woods.

The results discussed here underline the vital importance of the preservation of reference areas in which conditions and processes characteristic of a pristine state can be preserved (Tomiałojć *et al.* 1984, Tomiałojć & Wesołowski 1990, Angelstam *et al.* 1997, Tomiałojć 2000). To ensure this, the whole Białowieża Forest should be preserved. Unfortunately, the BNP protects only *c*15% of the Polish part, the remaining parts being commercially managed. Worse still is that logging is concentrated in the last remnants of old-growth stands of natural origin.

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**Ornis Hungarica 12-13: 63-72. 2003**

# Point count census using volunteers of terrestrial breeding birds in Norway, and its status after six years

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At present two monitoring programmes for terrestrial breeding birds are being undertaken in Norway. The Norwegian Directorate for Nature Management (DN) funds oneprogramme, which began in 1990, has paid fieldworkers. The other census programme, begun in 1995 and described here, is run by the Norwegian Ornithological Society in co-operation with Nord-Trondelag University College (HiNT). Its fieldwork involves unpaid volunteers who choose their own routes, each of 20 points. The number of participants has increased but slowly from the start so that in 2000 just 69 routes were investigated. These routes are unevenly distributed geographically, very few being in northern Norway. From the results, it is uncertain if the indices for the various bird species tell us only about changes in these routes, or if the data can be extrapolated to inform us about Norwegian populations. The paper discusses advantages and disadvantages of the census programme. The conclusion is that a new programme is needed and it is hoped to start in 2001. Because of the uneven distribution of volunteers and of the extremely difficult terrain, a semi-random approach will be applied. The country will be divided into regions, Of which five regions will be chosen and divided into 18km×18km squares. Within each chosen region 20 squares will be selected randomly. Each square will have 20 points determined according to a prescribed procedure. The information to be gathered at each point in the survey is discussed. This programme will be funded by DN, HiNT, and by companies sponsoring individual species. Participants in this new programme will have their expenses covered. Data on distribution and densities of Willow Warbler *Phylloscopus trochilus* and Chiffchaff *P. collybita* in Norway show exclusive competition between the species. The volunteer programme will continue.

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## 1. Introduction

The effect of human activity on environment has a great impact on living organisms. For example, it appears that since the 1950s, agricultural, forestry and construction has reduced the number of birds in wetlands, in forests or on farmlands throughout Europe (Tucker & Heath 1994). We modify the landscape and use increasing amounts and varieties of pesticides and other chemicals. Some of these chemicals may be very resistant to breakdown, thus staying in the food webs for a long time. Residues sometimes are still present in the food we eat. It is also known that some of the chemicals we release into the environment influence sexual development and can cause sterility. Pollution and large-scale landscape change may even be changing the climate. In general, increasing human populations and the scale of the application of technological developments have a massive and continuing negative impact on the environment. More than ever we need to monitor wildlife throughout Europe to see how natural populations are changing over time. The way populations change may give us important information about the health of our surroundings.

Birds are very well-suited as ecological indicators. They are relatively easy to study, and because many of the the public are knowledgeable about them, the work of amateurs can be harnessed in useful monitoring programmes. Because birds are relatively easy to identify, monitoring fieldwork methodology can be carried out with a high degree of confidence. Furthermore, birds have long been well-studied. Because species occupy a variety of niches throughout the food chain, some are likely to be particularly suitable for monitoring such as the accumulation of substances throughout that chain. Moreover, they may also be sensitive to some of the multiplicity of factors affecting the food chain. Those species with long life spans May well incorporate the effects of environmental stress over time, providing researchers with an opportunity to measure pollution over many years (Furness & Greenwood 1993).

Monitoring programmes must be effective and reliable. Well-defined objectives and a trustworthy methodology are necessary so that politicians and the public can accept the results and make the right decisions. The aims of the point count census programme of breeding birds in Norway, as designed by the Norwegian Ornithology Society, are to detect:

1. The impacts of pesticides and other pollutants on bird populations.
2. The effects of changing weather variables from year to year.
3. Still unknown threats to the environment.
4. The impact of human activity, especially land use, on bird populations.

To be able to do this, detailed information about population changes is needed.

To separate environmental factors from the masking effects of climatic changes, or to find any cause-effect relationships between birds and their environments, as many species as possible should be monitored (Koskimies & Väisänen 1991). It is also essential for a successful monitoring programme to be able to distinguish between natural and human-induced population changes, which may be difficult to do (Olsen *et al*. 1999). Because complete counts are out of the question for most bird species, careful selection of sampling design is essential. In the following I will present the methods used in the voluntary programme of point count census of terrestrial breeding birds in Norway, and discuss proposed methodological changes. Some findings are also presented.

## 2. Present monitoring system of terrestrial breeding birds in Norway

At present, two monitoring programmess for terrestrial breeding birds are being undertaken in Norway. The Norwegian Directorate for Nature Management started the first, a ‘Monitoring Programme for Terrestrial Eco-systems’ (MTE) in 1990. In brief, this programme includes monitoring of precipitation, soils, plant communities, birds (Golden Eagle *Aquila chrysaetos*, Gyrfalcon *Falco rusticolus*, Willow Grouse *Lagopus lagopus* and passerines) and mammals (mountain (Arctic) fox *Vulpes*

*(Alopex) lagopus*, mountain hare *Lepus timidus* and rodents) in seven permanent monitoring areas. In the MTE, monitoring is concentrated in the northern boreal and alpine ecosystems, and bird censuses are undertaken in the sub-alpine birch forest and above the tree line. In each of the seven areas the census takes place along 10 routes, each with 20 points. Stratified randomisation determines the placement of the routes. All participants in the MTE are paid.

The Norwegian Ornithological Society in co-operation with Nord-Trondelag University College (HiNT) runs the census programme (HFT) described below. After a couple of years of planning in co-operation with leaders of the Swedish (Svensson) and the Finnish (Väisänen) programmes, the census started in 1995. Some details of the census methods are:

**2.1. Selection of counting routes and points**

There are 20 points on each route. The participants can choose their route and points freely. The distance between points may vary, provided that there is a minimum of 350m between points in open areas and of 250m in forests, to minimise double-counting of individuals. Exactly the same points must be used in subsequent years for the population indices to have any value. The census must be taken by the same person each year. There are no restrictions on how the participants move between points (*e.g.* on foot or by a vehicle of any kind).

**2.3. Census periods**

The 'best' period for census-taking in southern Norway is from 10 May to 10 June, in central Norway from 10 May to 20 June, and in northern Norway from 30 May to 30 June. Although these are the recommended dates, because each routes is counted at the same time every year, the results for each route are comparable, no matter what the overall census period. For any route, timing of a census in later years should not differ by more than seven days from that of the first year.

**2.4. Time of day**

The best time for census taking is between 0400 and 0900. Census work is not allowed after 1000. The start of a count should not differ by more than 30 minutes from that of the first year.

**2.5. Weather**

Calm weather without precipitation is ideal. Point counts should be avoided if the weather is rainy or cold, or if the wind is moderate to strong. The census can be stopped and rescheduled to continue another day if the weather gets too bad.

**2.6. Field work**

The enumerator should approach the point carefully. The census period at each point is exactly five minutes. The surroundings within a 50m radius of the point are described and assigned a habitat code according to a prescribed list. Any habitat change from one year to the next within this 50m area is described. For each species observed (seen or heard), the number of pairs within and beyond the 100m circle is noted in a species list. An observed pair is defined as:

1. A male heard or seen.
2. A pair.

3. A single female.
4. A party of fledglings.
5. A nest.

Overflying birds are included as 'beyond 50m'. Flocks beyond 50m are registered in parentheses, (F5) meaning a flock of five birds. The organizing committee calculates the number of pairs in such flocks by taking into account the species, geography and time of year (Husby 1998). All bird species are counted.

In the Norwegian Ornithological Society, an organizing committee of three members manages the census programme. There is one contact person in each of the 20 counties. The most important functions for the contact person are to provide information about the census programme at meetings and to recruit qualified volunteers. Enumerators send completed census forms directly to the organizing committee after each season.

Svein Haftorn, a knowned Norwegian ornithologist, originally recommended this bird census programme (Haftorn 1995). Information about the project was presented orally at the annual meetings of The Norwegian Ornithological Society in 1994 and 1995. Subsequently, both general information and some of the results have been published annually in the Society's magazine. Information is also published in regional magazines, and an annual report is sent to the volunteers and other interested persons. One arbitrarily-chosen participant was given a bird book in 2000, and the same will be done this year.

All fieldwork is voluntary. The county contacts receive no remuneration. Members of the organizing committee do nearly all their work for free. The Directorate for Nature Management may provide future financial support for this monitoring programme, but that will depend on some methodological changes as explained below. Some economic support is received from Nord-Trondelag University College and The Norwegian Ornithological Society. From 2001, financial support will be provided by companies paying 5000 Nkr (almost 400 GBP) to sponsor individual species. So far of nine companies asked to sponsor species, the four that have agreed are; Norske Skog AS (Chiffchaff *Phylloscopus collybita*), Tronderenergi (Dipper *Cinclus cinclus*), Trondermat (Great Tit *Parus major*) and Aasen Sparebank (Magpie *Pica pica*).

Some results are presented, and an index is calculated for all bird species observed from a minimum of 20 routes. The index in the first year is set at 100, and the index the following years is calculated according to the formula:

$$\text{Index year 2} = \frac{\text{Index year 1} \times \text{Number of pairs observed year 2}}{\text{Number of pairs observed year 1}}$$

The routes have to be enumerated on two successive years by the same person before they are included in index calculations.

The collected data in the point count census can provide information to help determine relative densities in different parts of the country, habitat preferences and analyses of the competitive strength between different species. Chiffchaff and Willow Warbler *P. trochilus* are sympatric species in most of Norway, and they have a considerable overlap in both habitat and food selection (Saether 1983, Cramp 1992). Willow Warbler is the most common bird in the terrestrial bird monitoring programme in Norway, and in some parts there are also high densities of Chiffchaff. Is it possible that these two sibling species may competitively exclude one another at

Tab. 1. Indices for bird species observed on 20 routes or more. An index value of 100 is assigned to a species the first year the count exceeds threshold values. The sign indicates the significance level according to Spearman rank correlation between index values and the year, two-tailed test: * $P<0.05$, ** $P<0.01$ and *** $P<0.001$, and ns means not significant.

| | | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | Sign |
|---|---|---|---|---|---|---|---|---|
| Common Sandpiper | *Actitis hypoleucos* | | 100 | 100 | 109 | 152 | 118 | ns |
| Common Gull | *Larus canus* | | 100 | 114 | 125 | 127 | 113 | ns |
| Woodpigeon | *Columba palumbus* | | 100 | 90 | 112 | 123 | 99 | ns |
| Cuckoo | *Cuculus canorus* | | 100 | 113 | 174 | 136 | 159 | ns |
| Tree Pipit | *Anthus trvialis* | | 100 | 118 | 105 | 130 | 90,5 | ns |
| Pied Wagtail | *Motacilla alba* | | 100 | 82.8 | 96.8 | 114 | 109 | ns |
| Wren | *Troglodytes troglodytes* | | 100 | 77.7 | 121 | 151 | 145 | ns |
| Dunnock | *Prunella modularis* | | 100 | 90.4 | 88.1 | 91.1 | 88.2 | ns |
| Robin | *Erithacus rubecula* | | 100 | 77.8 | 75.2 | 116 | 109 | ns |
| Blackbird | *Turdus merula* | | 100 | 98.6 | 109 | 118 | 96.7 | ns |
| Fieldfare | *Turdus pilaris* | 100 | 118 | 119 | 147 | 144 | 168 | ** |
| Song Thrush | *Turdus philomelos* | | 100 | 87.3 | 125 | 126 | 134 | * |
| Redwing | *Turdus iliacus* | 100 | 102 | 118 | 126 | 141 | 141 | *** |
| Garden Warbler | *Sylvia borin* | | 100 | 96.7 | 115 | 115 | 121 | ns |
| Blackcap | *Sylvia atricapilla* | | 100 | 145 | 133 | 163 | 170 | * |
| Chiffchaff | *Phylloscopus collybita* | | 100 | 116 | 119 | 99.2 | 100 | ns |
| Willow Warbler | *Phylloscopus trochilus* | 100 | 98.1 | 101 | 117 | 112 | 102 | ns |
| Goldcrest | *Regulus regulus* | | 100 | 118 | 150 | 177 | 129 | ns |
| Spotted Flycatcher | *Muscicapa striata* | | 100 | 116 | 123 | 137 | 140 | *** |
| Pied Flycatcher | *Ficedula hypoleuca* | | 100 | 120 | 111 | 103 | 87.5 | ns |
| Willow Tit | *Parus montanus* | | 100 | 79 | 70.6 | 85.4 | 63.8 | ns |
| Blue Tit | *Parus caeruleus* | | 100 | 72.4 | 91.3 | 99.7 | 70.1 | ns |
| Great Tit | *Parus major* | | 100 | 97.1 | 99.1 | 95.4 | 85.2 | * |
| Magpie | *Pica pica* | | 100 | 123 | 139 | 172 | 148 | * |
| Carrion Crow | *Corvus corone* | | 100 | 126 | 141 | 146 | 143 | * |
| Starling | *Sturnus vulgaris* | | 100 | 116 | 114 | 147 | 160 | * |
| Chaffinch | *Fringilla coelebs* | | 100 | 95.3 | 94.1 | 92.5 | 92.4 | *** |
| Brambling | *Fringilla montifringilla* | | 100 | 117 | 90.2 | 113 | 97.8 | ns |
| Greenfinch | *Carduelis choloris* | | 100 | 120 | 120 | 105 | 151 | ns |
| Siskin | *Carduelis spinus* | | 100 | 129 | 114 | 97.3 | 115 | ns |
| Redpoll | *Carduelis flammea* | | 100 | 69.6 | 81.2 | 95.3 | 50.3 | ns |
| Yellowhammer | *Emberiza citrinella* | | 100 | 105 | 93.4 | 96 | 76.6 | ns |
| Reed Bunting | *Emberiza schoeniclus* | | 100 | 107 | 98.5 | 74.8 | 84.6 | ns |

the breeding grounds? The predictions are:

- If there is no competition between Chiffchaff and Willow Warbler, there will be low correlation values between the number of pairs of the two species in different parts of the country or in different habitats.
- If there is considerable competition between Chiffchaff and Willow Warbler, there will be a negative correlation between the number of pairs of the two species.

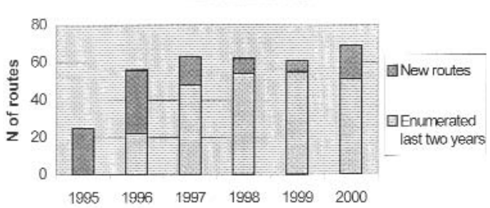


Fig. 1. Number of routes enumerated each year beginning in 1995.

## 3. Results

The number of participants in the bird census has increased slowly (the number of census routes is given in Fig. 1). In 2000,

Tab. 2. List of all habitat categories in which at least 20 points were enumerated. Habitat category numbers are the same as in Fig. 4.

| Habitat category number | Habitat category | Number of census points |
|---|---|---|
| 1 | Spruce forest without shrubs (spruce as dominant tree) | 49 |
| 2 | Spruce forest with shrubs (shrubs < 2m high) | 90 |
| 3 | Pine forest without shrubs | 54 |
| 4 | Pine forest with shrubs | 43 |
| 5 | Deciduous forest without shrubs | 21 |
| 6 | Deciduous forest with shrubs | 182 |
| 7 | Mixed forest without shrubs | 73 |
| 8 | Mixed forest with shrubs | 192 |
| 9 | Scrub with deciduous bushes and trees | 33 |
| 11 | Clear-cut area | 45 |
| 12 | Pine mire (a peat bog with some pines) | 29 |
| 13 | Open mire | 22 |
| 15 | Arable land (including grassland) | 82 |
| 17 | Pasture, possibly with scattered bushes or trees, or both | 37 |
| 18 | Rural settlement (buildings, yards, gardens etc) | 87 |
| 21 | Mountain birch forest | 36 |
| 25 | Habitat category other than above | 80 |
| 26 | Mixed habitat, the mixture comprising two or more habitat categories as given above | 136 |

57 different ornithologists enumerated the 69 routes. These routes are unevenly distributed throughout the country, as shown in Fig. 2. Most routes are enumerated well below the tree line, thus vertically separat-

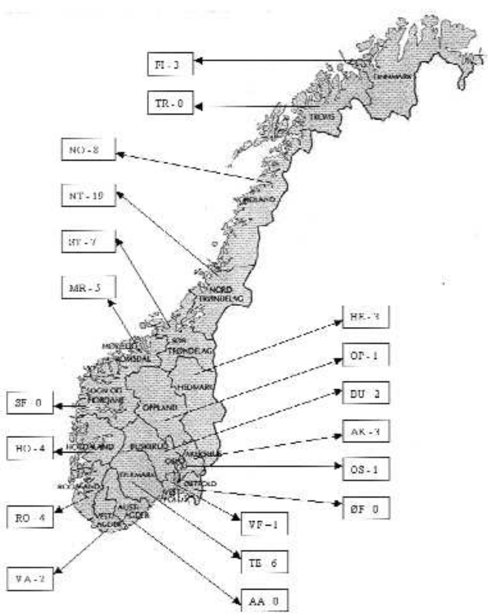


Fig. 2. Map of Norway showing the counties and the number of routes enumerated in each county in 2000.

ing this monitoring programme from the MTE programme.

During the last five or six years, there has been a significant increase in index values for birds that leave Norway and spend their winters in Europe (all species combined), especially Thrushes *Turdus spp*, Blackcaps *Sylvia atricapilla* and Starlings *Sturnus vulgaris*, but Chaffinches *Fringilla coelebs* have declined (Tab. 1). Index values for birds migrating to Africa were more variable, but the number of observations of Spotted Flycatcher *Muscicapa striata* has increased every year since 1996. Birds that winter in Norway also show variable trends: Tits *Parus spp* have declined, Corvids have increased, and finches vary with the seed production of various trees.

The relative densities of Chiffchaff and Willow Warbler in different counties in Norway are given in Fig. 3. The relative densities are given as the mean number of pairs observed in each route (20 points) in the various counties. The mean values are

Tab. 3. Spearman rank correlations between the number of pairs of Chiffchaff and the number of pairs of Willow Warbler in the two most popular habitats common to both species in 2000. Only points (n) where at least one of the species was observed are included.

| County | Deciduous forest with shrubs | | | Mixed forest with shrubs | | | Both habitats | | |
|---|---|---|---|---|---|---|---|---|---|
| | r | p | n | r | p | n | r | p | n |
| MR | -0.32 | >0.1 | 19 | -0.76 | <0.05 | 8 | -0.35 | <0.1 | 27 |
| ST | -0.57 | <0.1 | 10 | -1.0 | <0.001 | 9 | -0.73 | <0.001 | 19 |
| NT | -0.47 | <0.1 | 15 | -0.63 | <0.001 | 39 | -0.59 | <0.001 | 54 |
| MR-NT | -0.45 | <0.01 | 44 | -0.66 | <0.001 | 56 | -0.56 | <0.001 | 100 |

calculated for all counties with more than one route in the period 1996-2000. The eight counties in eastern Norway had significantly lower densities of both Chiffchaff (Mann-Whitney U-test, two-tailed: Z=-3.02, $P<0.01$) and Willow Warblers (Mann-Whitney U-test: Z=-2.31, $P<0.005$) than in the other eight counties.

The mean number of pairs of the two species at each point in different habitats is shown in Fig. 4. Naturally, the various habitats differ in importance for Chiffchaff and Willow Warbler. Mires and mountain birch forest hosted only Willow Warblers at significant densities. There is a weak positive correlation in the number of pairs of Chiffchaff and Willow Warbler observed in different habitats, though not significantly so (Spearman rank correlation: r=0.42, $P=0.08$). This means that both species have a tendency to prefer the same habitats, which increases the possibility of competition between them.

Looking at the eight counties in southern Norway and further northwards along the coastline, where the densities of both species were highest (Fig. 2), there were fewer Willow Warblers in areas where the densities of Chiffchaff were highest. This correlation was statistically significant (Spearman rank correlation: r=-0.83, $P=0.010$). To take a closer look at the competition between the species, I picked out the five habitats with the highest den-

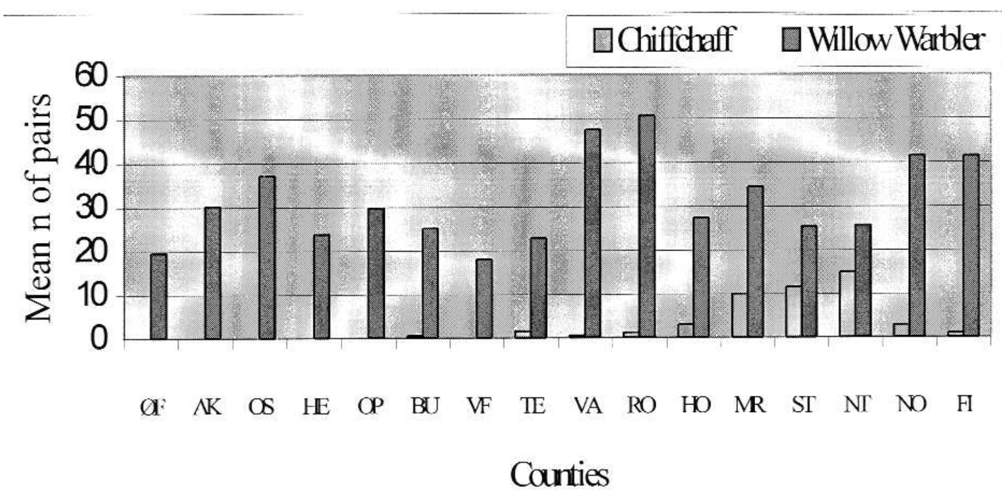


Fig. 3. Mean number of pairs of Chiffchaff and Willow Warbler registered at each point in the period 1996-2000. (Fig. 2 shows the location of the counties.)

sities of Chiffchaff, and the five habitats with the highest densities of Willow Warbler. Two of these habitats were common: a deciduous forest with shrubs (No 6 in Tab. 2 and Fig. 4) and a mixed forest with shrubs (No 8). The number of pairs of these two species were analysed to see if the number of pairs of one of the species was dependent or independent of the number of pairs of the other species. The analysis included only data collected within 50m of the census point, and only those points where at least one of the species was present. The three counties with the highest densities of Chiffchaff were included. The statistical tests are summarised in Tab. 3, and all three counties combined showed a strong negative correlation in each of the two habitats. The negative correlation was strongest in the two counties with the highest densities of Chiffchaff, and also in the habitat that contained most Chiffchaffs. The correlation became more negative by including only points with at least two pairs observed ($r=-0.79$, $P<0.001$, $n=34$) or at least three pairs observed ($r=-0.86$, $P<0.01$, $n=9$).

## 4. Discussion

According to Koskimies (1992), a national bird-monitoring programme must fulfil at least the following criteria. It must:

1. Be continual.
2. Be done in the same study areas from year to year.
3. Use comparable methods.
4. Cover as many species as possible.
5. Cover the whole country.
6. Cover all habitats, both optimal and marginal.
7. Detect both short-term and long-term population changes.
8. Be scientifically valid.
9. Have high efficiency.

Monitoring as many species as possible, as stressed by Koskimies & Väisänen (1991), allows us to separate the effects of some environmental factors from the masking effects of climatic changes, and to find any cause-effect relationships between birds and their environments. Both resident and migratory species must be included, because they experience different selection pressures during different seasons.

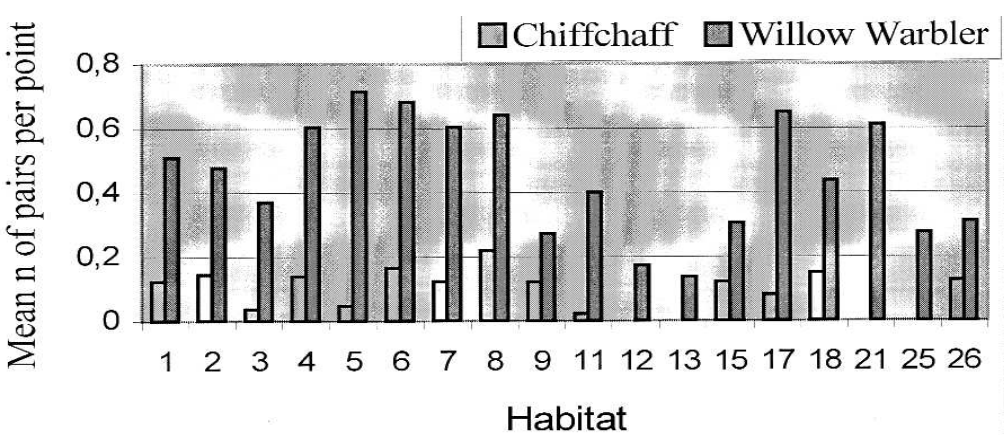


Fig. 4. Mean number of pairs of Chiffchaff and Willow Warbler observed in 2000 within 50m of each point in the various habitat categories. (Tab. 2 lists the habitat categories. Only those habitats in which at least 20 points were enumerated are included.)

Does the point count census programme of terrestrial breeding birds run by the Norwegian Ornithological Society in Norway fulfil these criteria? Fig. 2 shows the distribution of enumerated routes by volunteers in different counties, and this uneven distribution will always be a problem. We will not be able to enumerate the whole country (criterion 5 above) in a satisfactory way. Experience indicates that we will have the same distribution problem in the future.

Another weak point is that volunteers choose their routes and their points non-randomly. This is probably not justifiable. Thus the results probably cannot be generalised to infer what is happening to the population as a whole. It is important to be aware of what has been sampled and how it might relate to the whole population in relation to such factors as geographical distribution and habitat coverage (Bibby *et al*. 1992). Preliminary calculations in Sweden indicate that routes chosen non-randomly by volunteers contain more birds than randomly selected standard routes (Svensson *pers comm*). This implies that volunteers choose bird-rich habitats that are not representative for the whole country. As far as I know, no test of the relationship between index changes from volunteer non-random routes and random standard routes has been published, but both Britain and Sweden have used both systems for a few years and soon will be able to present correlations between the population indices derived from those systems. The results of these analyses will give some indication about the reliability of our method.

Whatever the results of these analyses, we will start selecting routes more randomly, and standardise the way to choose points along these routes. Not all details are clear at the moment, but a few alternatives suggest themselves.

The LUCAS grid system, of 18×18km squares, is applicable to the whole of Europe. A fully random choice of grids in Norway to enumerate will still be problematic because:

- It is a large country with few qualified inhabitants able to enumerate a representative number of routes in all regions (Its area is *c*324 000km$^2$, its population *c*4.5 million). Southern routes are liable to be well covered relative northern routes, where residents and participants are few.
- Much of Norway's terrain is unsuitable for bird census work, and so an arbitrarily chosen 20-point may include features such as large lakes, fast-flowing rivers, vast fjords and mountains that are both high and steep.

The plan is therefore to find a semi-random approach, which entails:

- The selection of about five regions in different parts of Norway and the collection of sufficient data from them to identify bird population and distribution trends.
- Each region, being 20 000-30 000km$^2$ in area, will include many LUCAS grids.
- All grids unsuitable for bird census work will be excluded.
- Among the remaining grids, at least 20 will be randomly chosen in each region.
- Participants will have their expenses covered and perhaps be paid wages, which is probably necessary if more than 100 routes in these five regions are to be enumerated.
- In each grid the census has to follow a

detailed standardised method. There are two possible methods discussed so far:

- Combine line transect and point counts along the 2×2km square centred on the grid centre. Every kilometer a point count will be undertaken, to a total of eight points. The line transect connecting the points will be enumerated. Each km should take between 30 and 40 minutes to complete.
- Make point counts all the way around the 2×2km square centred on the grid centre, with 400m between each point, making 20 points altogether.

  The first of these two approaches is used in Sweden. Preliminary calculations show slightly more observations per hour using line transect than using point count censuses, but the difference is small (Svensson, pers comm). Using point counts exclusively, the field method and treatment of the data will be uniform and similar to the existing programme.
- The current programme with volunteers choosing own routes will continue concurrently with the new proposed programme.

An essential aspect of any bird census is to discover reasons for changing bird populations. Therefore it is important to collect much information about the different factors affecting the birds. The Norwegian Institution of Land Inventory, The Norwegian Institute of Nature Research and other organizations are gathering various data in various regions, including data linked to the grid. Data will be collected on: pollution (local or remote in origin), precipitation, temperature, vegetation described by field investigation and by aerial photographic interpretation, other vegetation parameters (*e.g.* seed production, rate of growth), and some animal parameters (rodent population sizes, hare population changes). The bird population studies will be a part of an integrated study.

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**Ornis Hungarica 12-13: 75-87. 2003**

# Deriving population estimates for wintering wildfowl in Great Britain

M. Kershaw and P. A. Cranswick

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Information on the numbers of individuals in a population represents some of the most basic data that are needed to conserve populations effectively. Over the past decades, many wildfowl populations have undergone rapid changes in numbers as well as changes in distribution in response to the creation of refuges, management of populations, the creation of man-made wetlands and climate change. These continuing changes make it necessary to update population estimates on a regular basis. Data on the numbers of wildfowl wintering on wetland sites in Great Britain come primarily from the Wetland Bird Survey (WeBS), a joint scheme of the British Trust for Ornithology, the Wildfowl & Wetlands Trust, the Royal Society for the Protection of Birds and the Joint Nature Conservation Committee to monitor non-breeding waterbirds in the UK. Coordinated monthly counts by volunteers at wetland sites throughout Great Britain form the basis of the scheme, which focuses mainly on the months September through to March. It is not a simple task to calculate population sizes from extensive, volunteer-based surveys such as WeBS. In particular there are three main problems associated with the derivation of population estimates from WeBS data. Firstly, not all wetlands are covered by the scheme. Secondly, those that are covered do not represent a random selection of wetland sites. Thirdly, on any one count occasion there will be a number of missing counts from individual sites. In this paper we discuss methods for deriving population estimates for wintering wildfowl in Great Britain, by using WeBS data and evaluating past assessments of population sizes. A variety of different methods have been used to generate previous estimates and so it is important to distinguish whether a perceived change in population size is a real biological phenomenon or arises due to differences in the sampling method, the extrapolation method or the formula used to derive the estimate. The results of this analysis would suggest that previous population estimates have tended to underestimate the number of wintering wildfowl, and the resultant implications for conservation are discussed.

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## 1. Introduction

The Ramsar Convention on Wetlands of International Importance especially as Waterfowl Habitat, adopted in 1971, is the intergovernmental treaty that provides the framework for international cooperation for conservation and wise use of wetlands (Davis 1994). Contracting Parties are obliged to designate wetlands of international importance within their territory. The criteria for identifying wetlands of international importance were developed in 1974 and place particular emphasis on the importance of a site to waterfowl. Several criteria relate specifically to the numbers of waterfowl on a site. Those sites that regularly support 1% of the individuals in a population of one species or subspecies of waterfowl qualify as internationally important. The adoption of this 1% criterion has necessitated the generation of absolute measures of population size for waterfowl species and subspecies.

Generating an absolute measure of population size is not straightforward, especially for highly mobile, migratory species whose ranges span a number of political units. However, in northwest Europe waterfowl do tend to concentrate on discrete wetland sites during winter, thus offering the opportunity to survey a large proportion of the individuals of many species at one time. International population sizes are reviewed every three years (Rose & Stroud 1994, Rose & Scott 1997). In line with the international timetable, national population estimates for waterfowl wintering in Great Britain are scheduled for review every three years (Pollitt *et al*. 2000).

Great Britain is of outstanding importance for wintering waterfowl, over a million individuals being recorded in the peak winter months. In the UK, many Ramsar sites have been designated under the 1% criterion, which has also been used to identify wetland Special Protection Areas under the EU Birds Directive (79/409/EEC) (European Comission, 1979). Although Great Britain does not hold the total biogeographic population of most waterfowl species, sites which regularly hold 1% of the British population can be designated as nationally important and qualify for Sites of Special Scientific Interest (SSSI) status under the Wildlife and Countryside Act (Wildlife and Countryside Act 1981).

In this paper, we review past methods for calculating population estimates for wintering waterfowl in Great Britain and develop new methods for generating population estimates, which have been used to produce the most recent suite of population estimates for waterfowl in Great Britain (Kershaw & Cranswick 2003). In particular we look at the use of data from an extensive volunteer waterbird survey (WeBS) for generating population estimates.

## 2. Methods

Before it is possible to calculate population sizes it is necessary to define the term 'population'. This is relatively straightforward when the total biological population of a species is being considered, in which case the population size can be defined as the sum total of all the individuals in the population at a given time. A coordinated count of all locations where the population occurs would give the total population size at any one time.

However, when attempting to measure the number of individuals in a political unit (for example Great Britain), a coordinated count of all locations might represent only a fraction of the total numbers of individuals using the country during a winter season. There may be considerable turnover of individuals as birds move into and out of the political unit over the winter. Furthermore, there may be regional differences in the timing of arrival and departure of individuals. For these reasons national population estimates based on count data can represent only the peak number of individuals present in the region during the course of a winter and indicate only a minimum proportion of the species' biological population that uses the country.

Tab. 1. Estimates derived using the three methods used in the previous assessment of population size (see methods for details). The figure used to extrapolate the WeBS counts for each species is given in brackets in the last column. The overall population estimate is calculated as the mean of these figures for each species (see Tab. 3).

| Species | Peak Sum Monthly Means | Mean of Estimated Peak Count | Extrapolated Estimate (extrapolation figure) |
|---|---|---|---|
| Little Grebe *Tachybaptus ruficollis* | 4623 | 4387 | 5360 (1.43) |
| Great Crested Grebe *Podiceps cristatus* | 9970 | 9777 | 12494 (1.4) |
| Cormorant *Phalacrocorax carbo* | 17410 | 16158 | 18311 (1.24) |
| Mute Swan *Cygnus olor* | 20549 | 19031 | 28817 (1.65) |
| Bewick's Swan *Cygnus columbianus* | 6405 | 6004 | - |
| Whooper Swan *Cygnus cygnus* | 4613 | 3525 | - |
| European White-fronted Goose *Anser albifrons albifrons* | 5499 | 5419 | - |
| Greylag Goose (naturalised) *Anser anser* | 19365 | 19494 | - |
| Canada Goose *Branta canadensis* | 49956 | 49931 | 65605 (1.55) |
| Dark-bellied Brent Goose *Branta bernicla bernicla* | 90678 | 88857 | - |
| Shelduck *Tadorna tadorna* | 73312 | 72587 | - |
| Wigeon *Anas penelope* | 381853 | 372037 | - |
| Gadwall *Anas strepera* | 13857 | 12055 | 14280 (1.16) |
| Teal *Anas crecca* | 145369 | 138736 | 161739 (1.21) |
| Mallard *Anas platyrhynchos* | 184550 | 174197 | 259441 (1.71) |
| Pintail *Anas acuta* | 23701 | 23512 | - |
| Shoveler *Anas clypeata* | 10868 | 10083 | 12437 (1.22) |
| Pochard *Aythya ferina* | 47682 | 41869 | 49204 (1.18) |
| Tufted Duck *Aythya fuligula* | 62903 | 59632 | 70405 (1.32) |
| Goldeneye *Bucephala clangula* | 19123 | 18368 | 20282 (1.21) |
| Smew *Mergellus albellus* | 323 | 304 | - |
| Red-breasted Merganser *Mergus serrator* | 4730 | 4745 | 5296 (1.18) |
| Goosander *Mergus merganser* | 5203 | 4656 | 11329 (2.62) |
| Ruddy Duck *Oxyura jamaicensis* | 3625 | 3588 | - |
| Coot *Fulica atra* | 124756 | 109539 | 140462 (1.3) |

- no extrapolation

## Data

Data on the numbers of waterfowl wintering on wetland sites in Great Britain come primarily from the Wetland Bird Survey (WeBS), a scheme to monitor non-breeding waterbirds in the U.K. The main aims of WeBS are to:

1. Obtain population sizes.
2. Identify important sites.
3. Monitor trends in numbers and distribution.
4. Conduct research into population dynamics and ecology of waterfowl.

WeBS core counts are made at around 2000 wetland sites (3500 count units) of all habitats each year, coordinated counts at each site being made mainly between September and March. WeBS incorporates long-term count data that date back to 1947 for waterfowl. Data from 1960/61 have been computerised and comprise more than 400 000 visits to 8800 count areas, over 180 million wildfowl having been counted. Approximately 3000 volunteer counters contribute annually to the scheme, making 25 000 visits to WeBS sites each year. WeBS is not suitable for monitoring all species of wildfowl (*e.g.* seaducks) and so this paper considers methods for generating population estimates only for those species that are reasonably well represented by WeBS (Tab. 1).

When attempting to use WeBS data to

derive population estimates, there are a number of factors that affect the accuracy and representativeness of the estimates and the methods that can be used. Firstly, WeBS is a volunteer scheme and the sites counted are largely those that the counters choose to cover. For example, it is not possible to dictate which sites are counted or to assign a random selection of sites to counters, and so the main sites holding the largest number of waterfowl tend to receive good coverage, whereas smaller wetlands (or those habitat types that hold lower numbers of birds) tend to be underrepresented or omitted.

There are also differences in coverage relating to habitat and region. In general, estuaries, reservoirs and gravel pits are well covered whereas linear waterways such as rivers and canals are poorly represented relative to their extent in Great Britain. There are also biases in WeBS coverage relating to geographical region, with remote areas and those of low population density having lower coverage. Furthermore, we do not know what proportion of the total wetland resource in Great Britain receives coverage, making it difficult to extrapolate results from WeBS sites to all wetlands. Other biases relate to the nature of the data collected by WeBS, in particular the presence of missing counts from individual sites throughout the data series and incomplete counts from complex sites (*e.g.* large estuaries that comprise a number of individual count units).

The major advantage of this type of extensive volunteer-based survey is that a large proportion of the waterfowl present in Great Britain at any one time are probably counted. The major disadvantage is that it cannot be treated as a random sample and extrapolated to the total resource to produce an estimate. Additionally, there is the problem of how to deal with missing and incomplete counts within the scheme's sites.

Waterfowl tend to be highly mobile in winter, moving to other sites in response to factors such as cold weather and changes in water levels and in food resources. Numbers on sites can also fluctuate substantially from year to year. Traditionally, methods for assessing population size have used a window spanning (typically) five years to dampen short-term fluctuations in numbers. The previous assessment of population size for waterfowl in Great Britain covered the five-year period 1987/88 to 1991/92 and introduced a new methodology for calculating population sizes (Kirby 1995). The new methods recognised not only the need to account for missing values within the normal WeBS count but also that WeBS does not cover the whole of the wetland resource in Great Britain. For those species where WeBS achieves a reasonable level of coverage, three different methods were used and the mean of these was taken to produce a population estimate (Kirby 1995, Stone *et al*. 1997). Two of these methods used only WeBS count data, but attempted to take into account that on any count occasion, there will be counts missing from some of the WeBS sites. The third method recognised that WeBS sites do not represent all wetlands and so even if all WeBS counts were complete, a certain proportion of the population would still go uncounted.

Two methods were employed to account for the problem of missing values in the WeBS dataset (Kirby 1995):

1. Peak sum of monthly means. This method took the mean site count for

each month over the five-year period and then summed this value across all sites. The population estimate selected is then that for the month with the highest value.

2. Mean of estimated peak counts. This method involved selecting the month of peak abundance for each species (derived from index values; see Kirby [1995]). If the count for a site in the peak month was missing, then the count from the next available month in that year was selected, or if none of the alternative counts was available, then a count from the preceding year was selected. The population estimate was then calculated by summing the counts across all sites for each year and then taking the five-year mean.

To estimate the number of birds occurring outside WeBS sites, the previous analysis of population sizes used data from three regional intensive surveys, of northwest England, northeast England and southwest London (Quinn & Kirby 1995, WWT unpublished data). In these surveys an attempt was made to count every waterbody in the region concerned, enabling a calculation to be made of the number of birds on WeBS sites in the region, relative to the total regional population. The mean proportion of birds on WeBS sites relative to the regional total from these three intensive surveys was then used to correct the WeBS five year peak mean for each species nationally, to derive a total population estimate for Great Britain. The assumption was that the blitz survey regions were representative of all regions in terms of the relationship between birds on WeBS sites and total bird numbers.

The mean of the three different methods was used to generate a single population estimate for each species (Kirby 1995). These same methods were used to generate new population estimates for waterfowl wintering in Great Britain. The most recent data available were from the period September 1994 to March 1999 and included data from only the WeBS core winter months September through to March in each year.

However, there were several problems with the methods used in the final population assessment. The two methods for accounting for missing counts will tend to underestimate the true number of birds. Using the Peak sum of monthly means method, it is possible for a site to have no counts at all in a particular month in each of the five years and so the site will not feature in the population estimate. Furthermore, a site's mean count for a particular month could be based on just one or two counts, which could result in an under- or overestimate of the average five-year population size, especially if numbers have changed substantially over the five-year period. Moreover, this method assumes that the seasonal phenology is similar across years, but if the peak count occurs in different months across the five-year period, then averaging the monthly count across years, summing across sites and selecting the highest value will produce a figure that is lower than the method of taking the five-year peak mean value. Using the Mean of estimated peak counts method, there remains a number of sites where no counts are available to impute, either from another month in the same winter or from the previous winter (in other words some counts are still missing). Furthermore, even if a count can be substituted using a count from another month or year, this count will tend to be lower (because it is not from the month of peak abundance).

Neither of the above methods attempts to compensate for missing counts within complex sites: ie the situation where a site count is incomplete (site counts were treated as complete even if some units within them remained uncounted). Both methods will therefore always produce an underestimate of the number of birds on WeBS sites.

Extrapolation based on enhanced coverage on its own is likely to give the best indication of national population size. However, the intensive surveys were restricted in terms of geographical region or in the case of the southwest London survey, habitat type covered. The northwest England survey was carried out during a period of cold weather when birds were likely to be more concentrated (on to the bigger, and in many cases, WeBS sites). Additionally, in calculating the relationship between birds on WeBS sites and total birds, the extrapolation figure was derived by comparing the surveyed birds that were on WeBS sites with the total counted. However, this assumes complete coverage of the WeBS sites, whereas on a typical WeBS count occasion some of the WeBS sites will go uncounted. Therefore the extrapolation should only be applied to the WeBS count, after accounting for missing values.

The population estimates generated by the two methods that accounted for missing values in WeBS counts give a population estimate only for birds on WeBS sites. For some species like Pintail *Anas acuta* this will be close to the national population size, but for other species like Mallard *A. platyrhynchos* the figure will be much lower. The extrapolated counts based on enhanced coverage surveys give an indication of the national population estimate, across all sites, not just WeBS sites. For this reason it is not valid to calculate a mean across all three methods. A mean value based just on WeBS counts is valid to produce an estimate of the number of birds on WeBS sites (although it might be better to select the maximum value since we know that the methods are more likely to underestimate numbers), but this should not be treated as comparable with the extrapolation method based on intensive coverage which produces a national estimate of population size across all wetland sites.

Despite these methodological problems, WeBS data remain the best available data on the numbers and distribution of the majority of waterfowl species wintering in Great Britain. However, there is a need to account for missing counts within the WeBS database in a more comprehensive manner than in the previous population assessment. It is necessary also to be able to extrapolate from WeBS sites to the total wetlands resource in order to generate a national population estimate, but WeBS estimates should not be combined with the extrapolated estimates. For this assessment of British population sizes, new methods were used which aim to address the above deficiencies.

In an attempt to account for missing counts in the data, an indexing technique was used to impute values where a count was missing using a simple multiplicative model having site, year and month factors (Underhill 1989, Underhill & Prŷs-Jones 1994). Indexing was also used to compensate for the effect of incomplete site counts when some sectors of complex

Tab. 2. Comparison of the % of missing counts imputed by the mean of estimated peak count and the index model methods. The "% missing after estimation" gives the % of the potential, complete dataset that remains missing after the addition of available estimated counts by the mean of estimated peak counts method. The "% of total count imputed" gives the % of the total number of birds that consists of estimated counts. The "% of actual counts imputed" gives the % of the available counts (correct plus imputed) that were estimated counts. Under the index model method, the first column is the total % of counts that were missing or incomplete 94/95 - 98/99 and the second column gives the % of birds that were imputed.

| Species | Mean of Estimated Peak Method | | | Imputing using Index Model | |
|---|---|---|---|---|---|
| | % missing after estimation | % of total count imputed | % of actual counts imputed | % counts missing or incomplete | % total bird numbers imputed |
| Little Grebe | 14.7 | 20.3 | 35.2 | 33.9 | 28.1 |
| Great Crested Grebe | 13.2 | 17.4 | 32.1 | 32 | 21.2 |
| Cormorant | 15.0 | 11.6 | 20.9 | 34.5 | 25.2 |
| Mute Swan | 17.5 | 25.8 | 37.6 | 37.6 | 26.6 |
| Bewick's Swan | 6.48 | 0.900 | 9.77 | 24.4 | 10.9 |
| Whooper Swan | 13.2 | 14.1 | 16.5 | 32.6 | 24.2 |
| European White-fronted Goose | 5.80 | 0.148 | 6.88 | 21.4 | 4.5 |
| Greylag Goose naturalised | 11.2 | 24.5 | 32.0 | 29.7 | 25.8 |
| Canada Goose | 15.6 | 22.6 | 34.8 | 33.9 | 28 |
| Dark-bellied Brent Goose | 3.67 | 0.518 | 5.78 | 18.2 | 10.7 |
| Shelduck | 11.7 | 1.17 | 12.8 | 29.4 | 15.3 |
| Wigeon | 13.4 | 2.83 | 15.9 | 32.7 | 15.8 |
| Gadwall | 10.7 | 6.52 | 13.8 | 27.6 | 20.7 |
| Teal | 15.1 | 5.32 | 19.6 | 34.9 | 23 |
| Mallard | 20.7 | 14.2 | 24.0 | 45 | 30.6 |
| Pintail | 8.07 | 2.48 | 11.2 | 24 | 13.2 |
| Shoveler | 10.0 | 5.85 | 16.2 | 27.1 | 21.3 |
| Pochard | 13.8 | 10.3 | 18.9 | 33.3 | 22.6 |
| Tufted Duck | 16.9 | 11.8 | 20.6 | 36.9 | 26.3 |
| Goldeneye | 14.8 | 9.31 | 18.0 | 34.8 | 22.8 |
| Smew | 4.90 | 3.68 | 9.05 | 21 | 20.3 |
| Red-breasted Merganser | 14.4 | 5.83 | 16.8 | 34.5 | 23.6 |
| Goosander | 13.3 | 12.3 | 15.1 | 31.4 | 32.8 |
| Ruddy Duck | 8.07 | 9.40 | 13.3 | 24.5 | 15.5 |
| Coot | 17.1 | 13.0 | 22.8 | 37.1 | 25.4 |

sites went uncounted on a particular occasion. Here, a count was flagged as potentially incomplete if less than 75% of the total sectors and less than 75% of the maximum bird numbers for a particular month had been recorded. The imputed value from the indexing model, where this was greater than the incomplete count recorded, was then substituted for the actual count. Otherwise, the actual count was retained. The five-year peak mean count for each species was then calculated using the real count where it existed and imputed values where counts were missing or incomplete. This figure gives an estimate of the total number of individuals on WeBS sites.

In order to generate an estimate of the national population size, it is necessary to extrapolate the WeBS total according to the proportion of the population that WeBS covers. This is largely a matter of conjecture since it is not known what proportion of populations, or indeed wetland sites, that WeBS covers. The best indication of the proportion of birds on WeBS sites comes from regional intensive surveys carried out in the early 1990s.

Tab. 3. Population estimates derived using alternative methods. The first two columns give the minimum and maximum population estimates on WeBS sites for each population. The minimum population estimates represent the highest value out of 1) the peak sum of monthly means, 2) the mean of the estimated peak count and 3) the five year peak mean. The maximum estimated population represents the five year peak mean, calculated with all missing counts imputed according to an index model. The third column gives the population estimates that would be derived using the previous methods of (Kirby 1995). The final column is the estimate derived using the new methods. All values rounded to the nearest 1000 for estimates >100,000, the nearest 100 for estimates 10,001-100,000, the nearest 10 for estimates >1001-10,000 and the nearest one for estimates <1001.

| Species | Minimum WeBS | Maximum WeBS | Population using previous methods | Population using new methods |
|---|---|---|---|---|
| Little Grebe | 4,620 | 5,430 | 4,790 | 7,770 |
| Great Crested Grebe | 9,970 | 11,400 | 10,700 | 15,900 |
| Cormorant | 17,400 | 18,600 | 17,300 | 23,000 |
| Mute Swan | 20,500 | 22,700 | 22,800 | 37,500 |
| Bewick's Swan | 7,180 | 8,070 | 6,210 | 8,070 |
| Whooper Swan | 4,610 | 4,850 | 4,070 | 5,720[a] |
| European White-fronted Goose | 5,600 | 5,790 | 5,460 | 5,790 |
| Greylag Goose (naturalised) | 19,500 | 22,900 | 19,400 | 28,500[a] |
| Canada Goose | 50,000 | 62,000 | 55,200 | 96,100 |
| Dark-bellied Brent Goose | 95,900 | 98,100 | 89,800 | 98,100 |
| Shelduck | 73,300 | 78,200 | 73,000 | 78,200 |
| Wigeon | 382,000 | 406,000 | 377,000 | 406,000 |
| Gadwall | 13,900 | 14,700 | 13,400 | 17,100 |
| Teal | 145,000 | 159,000 | 149,000 | 192,000 |
| Mallard | 185,000 | 206,000 | 206,000 | 352,000 |
| Pintail | 25,600 | 27,900 | 23,600 | 27,900 |
| Shoveler | 10,900 | 12,100 | 11,100 | 14,800 |
| Pochard | 47,700 | 50,500 | 46,300 | 59,500 |
| Tufted Duck | 62,900 | 68,300 | 64,300 | 90,100 |
| Goldeneye | 19,100 | 20,600 | 19,300 | 24,900 |
| Smew | 323 | 370 | 314 | 370 |
| Red-breasted Merganser | 4,750 | 5,520 | 4,920 | 6,510[b] |
| Goosander | 5,200 | 6,140 | 7,060 | 16,100 |
| Ruddy Duck | 3,630 | 4,110 | 3,610 | 4,110 |
| Coot | 125,000 | 133,000 | 125,000 | 173,000 |

[a] New population estimate calculated using an extrapolation figure derived from additional census data (see Kershaw & Cranswick 2003)

[b] This estimate derived from WeBS data was not the final adopted population estimate for Red-breasted Merganser (see Kershaw & Cranswick 2003)

Unfortunately there have been no intensive surveys since, so in the current assessment of population size the extrapolation figures from in the previous assessment (Tab. 1) were used. The five-year peak mean, calculated with all the missing and incomplete values imputed using an index model, was multiplied by the extrapolation figure to generate an estimate of the national population size for each species.

## 3. Results

During the five-year period 1994/95 to 1998/99, a total of 2773 wetland sites (comprising 4328 individual count units) was counted during the winter period September through to March. More than 30% of these sites had five or fewer missing counts out of the potential thirty-five for the period, although 15% of sites had more than thirty missing counts. However, the sites with the most complete coverage were also the numerically most important for waterbirds, holding, on average, almost five times the number of birds compared to sites with more than thirty counts missing. This means that the percentage of the count missing for most species was in fact much lower than the overall coverage achieved would suggest.

At a species level, if only sites where a species was present between 1994/95 and 1998/99 are included, between 18 and 45% of site level counts were missing or incomplete (Tab. 2). Those species that have restricted distributions (and so occur on relatively few sites), such as Bewick's Swan *Cygnus columbianus* and Dark-bellied Brent Goose *Branta bernicla bernicla*, and those that show highly aggregated distributions, such as Pintail, tended to have the lowest percentage of counts missing or incomplete. In contrast, widely distributed, dispersed species like Mallard and Mute Swan *Cygnus olor* have a higher percentage of counts missing or incomplete because they are more likely to occur on sites that are poorly covered.

The population estimates derived using the three methods used in the previous assessment are given in Tab. 1 and the mean population estimate is given in Tab. 3. For the majority of species the Mean of the estimated peak count produced an estimate that was on average 6% lower compared to the Peak sum of monthly means. Tab. 2 shows the percentage of potential counts that consisted of estimated values using the Mean of estimated peak counts method to impute missing values, and the percentage of counts that remained missing after the estimation procedure had been applied. Between 4 and 21% of counts remained missing after the addition of available counts from months outside the peak month, depending on the species. The percentage of the total count that consisted of imputed values was always substantially lower than the percentage of counts that these imputed values represented. For example, 16% of Wigeon *Anas penelope* counts were imputed, but these only represented 3% of the birds. Although the missing counts came disproportionately from the less important sites (so it would be expected that the counts would be of lower magnitude), coupled with the missing counts remaining after imputing, the Mean of estimated peak counts method of accounting for missing counts will tend to produce an underestimate of the total number of birds.

Since the methods used in the previous assessment of population sizes to account for missing counts were considered likely to produce underestimates, an alternative method was used to impute missing values. An index model was used to produce estimated counts based on a 'site, year and month' factor and these estimated counts were used to fill the missing values. Tab. 2 shows the percentage of counts missing and incomplete and the percentage of the total bird numbers that represent imputed values for each species for the whole period 1994/95 to 1998/99. Most species had between 20 and 35% of counts missing or incomplete, but the percentage of the total count that this imputed element represented varied by more, partly dependent on factors such as how dispersed or widespread the species was. For example, 33% of Wigeon counts were missing or incomplete, but only 16% of the count was imputed, because the species tends to be aggregated on well-covered sites, so that most of the imputed counts were of small numbers on the less important, less well-covered sites. In contrast, 34% of Little Grebe *Tachybaptus ruficollis* counts and 32% of Goosander *Mergus merganser* counts were missing or incomplete and imputed values represented 28% and 33% of the total count respectively. Both these species are more dispersed than Wigeon.

The effect that the imputed counts had on the population estimate for WeBS sites varied according to the species. Tab. 3 compares the maximum value that could have been derived using the previous methods for assessing population size on WeBS sites (*ie* the maximum value from the Peak sum of monthly means, the Mean of the estimated peak count and the five-year peak mean) with the five-year peak mean calculated when all the missing and incomplete counts are imputed using an index model (Columns 1 and 2 in Tab. 3. respectively). The difference between these two values varied from 2.3% (Dark-bellied Brent Goose) to 24% (Canada Goose *Branta canadensis*). The largest differences were for dispersed species where the cumulative effect of missing counts over a large number of sites (including the smaller less well-covered sites and poorly covered habitats such as rivers) adds up to a significant number of birds missed. Such species include Goosander (18% difference between lower and upper estimates), Little Grebe (18%) and naturalised Greylag Goose *Anser anser* (17%). The smallest differences were for species which tend to be concentrated on the larger, well-covered sites, for example, European White-fronted Goose *Anser albifrons* (3.4%), Whooper Swan *Cygnus cygnus* (5.2%) and Wigeon (6.2%).

The estimates in columns one and two of Tab. 3 indicate only the estimated peak number of birds on WeBS sites. To generate a national population total requires extrapolation from WeBS sites to all sites in Great Britain. The figures used to extrapolate WeBS counts to produce a national population estimate were based on data from three regional blitz surveys carried out in the early 1990s, taken from Kirby (1995) (see Tab. 1). The figures represent the proportional increase (%) in the numbers of birds counted during the blitz survey compared to the number of birds that were present on the WeBS sites within the survey region. The mean value from the three intensive surveys was used to extrapolate the five-year peak mean WeBS count (with all missing values imputed [column 2 of Tab. 3]) (Tab. 3). For naturalised Greylag Goose and Whooper Swan, extrapolation figures were calculated from national surveys that had been carried out for the species (see Kershaw & Cranswick 2003 for details). The extrapolated population estimates represent the estimated peak number of birds wintering in Great Britain and are between 6% and 128% higher than the estimates calculated using the previous methodology (Tab. 3).

## 4. Discussion

One of the earliest assessments of the number of waterfowl wintering in Great Britain used average January waterfowl census data over the period 1967-73 as a measure of population size (Atkinson-Willes 1976). This method took no account of missing counts within the time period on sites within the census or of sites that were not included in the census at all. In the mid-1980s, a method was used to account for both missing counts in census sites and for sites not covered at all (Owen *et al*. 1986). To account for missing counts in the census sites, the five-year peak mean for a species was compared to the peak monthly mean summed across all sites. Using this method Owen *et al*. (1986) calculated that annually WeBS would cover, for example, 70% of the Pintail on WeBS sites, 76% of Wigeon, 89% of Gadwall *Anas strepera*, 73% of Mallard, 75% of Pochard *Aythya ferina* and 80% of Tufted Duck *Aythya fuligula*. These percentages enabled the peak annual count for each species to be corrected to compensate for sites not counted. This total was then corrected for sites completely missed in the five-year period (Owen *et al*. 1986). This correction was based on a 'best guess' such that, for example, 10% of Tufted Duck and Pochard and 50+% Mallard were estimated to occur on non-WeBS sites.

The previous assessment of wintering waterfowl population sizes in Great Britain (1987-1991) applied a new methodology to the problem of generating population estimates from an extensive, volunteer-based survey (Kirby 1995). However as the current analysis illustrates, these methods are likely to underestimate substantially the peak numbers of waterfowl occurring in Great Britain during the winter. In particular, insufficient accounting for missing values and calculating a mean population size using both the WeBS and national estimates will result in an underestimate of true numbers.

Accounting for missing counts using an index model and using the extrapolated totals on their own as an indication of national population totals results in some substantial increases in population size. The new methods result in population estimates that are between 6% (European White-fronted Goose) and 128% (Goosander) higher than using the previous methods. The largest increases are for those species that are widely dispersed across sites, or where a significant proportion of the individuals are on habitats that are poorly covered by WeBS. For example, compare Mallard (71%), Mute Swan (64%) and Little Grebe (62%), with more aggregated species like Wigeon (8%), Shelduck (7%) and Pintail (18%).

This result has implications in terms of sites that qualify as important. Applying the new estimates of national population size to generate national 1% levels means that almost all species would have fewer sites qualifying as nationally important. The most marked change occurs for Gadwall where the number of nationally important sites would fall from 82 sites to 27 compared to a fall from 37 sites to 25 for Wigeon. However, Mallard would show an increase from zero sites to one (based on 1998/99 data in Pollitt *et al.* 2000).

Despite the large differences between the methods used for the last assessment and the most recent, applying the same methodology as Owen *et al*. (1986) gives comparable results to the new methods presented here. For example, Pochard 59 300 (61 775 Owen), Mallard 352 000 (415 671, although Owen *et al*. used 500 000). The population sizes derived using the figures of Owen *et al*. (1986) in fact tend to be slightly higher than those presented in this paper. This might be expected, however, since WeBS coverage has improved since the late 1970s, both in terms of the coverage within WeBS sites and coverage of wetland resource in Great Britain, and so the extrapolation figures used by Owen *et al*. (1986) are now probably slightly high.

All these estimates are considerably lower than the total numbers of each species using Great Britain during each winter season. For Teal *Anas crecca*, the estimated hunting kill was 180 000 per annum 1979-81 (although Bertelsen & Simonsen [1989] quote a kill of 288 000 Teal for Great Britain and Northern Ireland annually) compared to a peak winter count in the region of 100 000 at the time of the bag estimate (Owen *et al*. 1986). Similarly, the estimated annual kill for Mallard was 600 000 compared to a population estimate of 500 000 and a peak winter count of 190 000 (Owen *et al*. 1986).

The need for an absolute measure of population size arises largely as a result of the 1% criterion (Atkinson-Willes 1976). Under this criterion sites that hold 1% of a population qualify as internationally important under the Ramsar Convention, and more recently the 1% criterion has been used for designating SPAs (Special Protected Area) under the EU Birds Directive. At a national level, sites in Great Britain that support 1% of the national population of a species can be designated as Sites of Special Scientific Interest. Although the 1% criterion has no biological basis (for example, sites with 1% of a population are not known to be viable compared with sites holding less than 1%) application of this criterion requires an estimate of absolute population size. This allows a major criticism of the 1% criterion, that it is dependent on an estimate of absolute population size, something that is virtually impossible to measure, even in cases where there are very good survey data.

Despite the problems associated with generating population estimates, it is still valuable to be able to extrapolate the numbers on WeBS sites to numbers at a national level. In order to extrapolate more accurately, it is essential that new intensive surveys are carried out in Great Britain to determine the relationship between birds on WeBS sites versus the total wetland resource. Since WeBS sites tend to represent the larger more important wetlands, it is possible that if populations increase, numbers on WeBS sites may reach carrying capacity and birds will move on to less important sites which may not be counted. Conversely, management of the well-monitored sites for waterfowl populations may result in numbers faring well on these sites, masking detrimental changes in population status on smaller non-WeBS sites or the wider countryside. A major criticism of the above-mentioned methods is their inability to calculate the precision and accuracy for the population estimates. We know that the estimates are more accurate for highly clumped species like Wigeon, which occur on the larger, better-covered sites. However, we cannot quantify the levels of precision or accuracy of the estimates. One approach might be to focus counters' effort on achieving complete coverage of fewer sites so as to minimise the effect of missing counts in the data series. There is considerable scope for more work to determine the best way of using data collected from this extensive volunteer-based survey to calculate population sizes for waterfowl in Great Britain.

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**Ornis Hungarica 12-13: 89-94. 2003**

# Monitoring of breeding water birds in Lithuania: organisation and sampling designs

V. Stanevičius

Stanevičius, V. 2003. Monitoring of breeding water birds in Lithuania: organisation and sampling designs. – Ornis Hung. 12-13: 89-94.

The first Lithuanian national programme of monitoring breeding water birds was launched in 1999, although in 2000 year some parts had to be suspended, but from 2001 it will be continuous again. The programme is funded by the Ministry of Environment Protection of Lithuania and is coordinated by the Institute of Ecology. The subjects of sampling methods and of project organization were studied and evaluated with particular care before the work began. Planning was affected both by the limited funds and the non-uniform distribution of skilled observers. A network of 78 monitoring points was set up in 19 of the 43 districts. Despite the fact that a random approach could not be applied to the sampling design, the sample reflects the geographical distribution and represents the ecological variety of Lithuanian wetlands. The area covered by the monitoring points comprises 37 lakes of 7 ecological types, two fish pond complexes, parts of two water reservoirs , 32 gravel and clay banks and pits, and sections of 5 rivers, large and small. The areas are situated in relatively natural or urban landscape. The majority of the 18 observers worked in National and Regional parks, or in Strict Regime reserves. They gathered data on the numbers and distribution of 29 common and rare waterbird species.

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## 1. Introduction

The original purpose of the Lithuanian programme of monitoring breeding water birds was to identify long-term trends in the development of waterbirds, for such data are required to plan and realize more effective wetland avifauna protection and relevant management measures than had been possible previously. Cardinal changes in the waterbird fauna community and in the environment itself occurred in Lithuania since the 1950s. The most important of these changes include:

1. Rapid intensification of agriculture.
2. Hyper-eutrophication of traditional 'bird-lakes' resulting in drastic declines in number of the mass of waterfowl species (Stanevičius 1999).
3. Large-scale landscape and agricultural melioration policies introduced from the 1960s onwards,.
4. Regulation of lake water levels.
5. Cessation of grazing and hay-making on lake islands and in lake-side meadows (resulting in the degradation of these important habitats).
6. Systematic development of large fish-pond complexes (and the recent tendency for them to go out of use) (Švažas & Stanevičius 1999).
7. Creation of several large water reservoirs and numerous smaller man-made water bodies.
8. Introduction of some mammalian predator species, such as American mink *Mustela vison* and raccoon

*Nyctereutes procyonoides*.

9. Reintroduction of Greylag Goose *Anser anser* and Cormorant *Phalacrocorax carbo*, which through persecution became extinct in Lithuania; their breeding populations are now increasing.

The registration of breeding species new to Lithuania, such as Great Egret *Egretta alba* is a possibility in the near future. The impact of global climate warming on waterbirds has already been identified in Lithuania (Žalakevičius 1998, 1999), and is a new and compelling argument for maintaining the monitoring of breeding waterbirds.

The Lithuanian programme of monitoring breeding waterbirds can be compared to other European national programs (see Gibbons 1999). Our programme was designed to take into account the circumstances arising from the enormous recent changes in Lithuanian society, which have had both subjective and objective effects. In time, the programme overlaps with the efforts to combine national programmes into the Pan-European programme (Gibbons 1999). Like other eastern European projects, the Lithuanian programme is affected greatly by lack of funds and the shortage and non-uniform distribution of skilled observers. The main obstacle to achieving a unified approach is the variety of earlier ways of counting waterbirds (and their variable methodology design and application), circumstances brought about by local and uncoordinated work in the past. Both national and European monitoring approaches have their own merits and shortcomings. Ignorance of local peculiarities is characteristic of imposed and non-systematic programmes, leading to loss of some valuable information (Gibbons 1999). On the other hand, lack of comparability of national data reduces their value. At the present stage of our monitoring work we would like to believe that some comparison of data is reliable (Strien & Pannekoek 1999). In the first year of such a project, organization and methodical aspects are of great importance (Szép & Gibbons 1999). The major purpose of the Lithuanian monitoring in its initial stage is to prepare and test the preliminary design of the methodology, which would be sufficient to evaluate the significance of developments in Lithuanian waterbird populations. To allow these aims to be achieved, we have tried to create a representative network of monitored sites, to perform waterbird counts on selected wetlands and to determine the primary characteristics of waterbirds in the sites being monitored.

The above aims mean that any elaboration of the study area selection criteria and of the count methods described in my paper will come from the feedback from my work. Consequently, the feedback and the modified criteria must be discussed in objective detail. The methodology and area selection criteria previously used and cited in scientific papers in Lithuania are inadequate for the work of the present monitoring programme and should now be discarded.

## 2. Methods

The practices employed by a number of authors were studied before selecting a method of implementing counts of breeding waterbirds (*e.g.* Borowiec *et al*. 1981, Haldin & Ullfvens 1987, Klett *et al*. 1986, Maxson *et al*. 1986-1987, Rumble & Flake 1982). The method reflected adaptations to cope with monitoring waterbirds in Lithuanian conditions. We also considered the experience we gained in the traditional year on year nest search and survey of all

Great Crested Grebes Podiceps cristatus and Coot Fulica atra at Lake Žuvintas. In the end, our counting methods of breeding waterbirds included:

1. Total nest searches.
2. Evaluation of breeding population size by:
   a. Voices.
   b. Pairs.
   c. Isolated mated males using waiting places.
   d. Broods.
   e. Species-specific breeding behaviour.

Such a diversity of methods used in counting waterbirds is practically inevitable (Borowiec *et al*. 1981). We discussed all the above methods with all the participants and used an appropriate selection to accord with the local characteristics of each wetland surveyed.

It is essential that counters use the same methods every year in the same wetlands, although counting methods may differ between sites. I recognize that the variety of methods could result in differences in precision of the estimations. Nevertheless, if the method-introduced error remains the same every year (the same method always being used on any particular site) it will not preclude the correct evaluation of the direction and magnitude of changes.

Counts of breeding water birds were performed every two weeks, commencing in early April and ending in mid-July, which achieved a total of 6-7 counts per monitoring site. Participants received a special count card. For each wetland, each year, a dedicated card recorded all (6-7) counts performed. After each count, additional data were added to the cards in the appropriate columns devoted to nests, groups comprised of paired birds, lone pairs, single males, single females, single unsexed birds, groups of <6 males and broods.

After the count season, completed cards were returned to the project coordinator in the Institute of Ecology (Bird Ecology Laboratory). Count data are stored in computer on Excel data sheets in two locations, the Institute of Ecology and the Ministry of Environmental Protection. There are plans to create a database.

## 3. Results and Discussion

### 3.1. Criteria of sampling designs

We attempted to select monitoring sites in such a way that we could identify changes in breeding water bird populations in time and space. To achieve this, when selecting monitoring sites we ensured that they were separated as far from each other as possible. To avoid false conclusions, we attempted to count birds in wetland types that differed ecologically, which is why the sample includes wetlands of different carrying capacity, from unproductive mire lakes in moss bogs to mesotrophic and eutrophic lakes of agricultural landscapes. Some of the sites were selected as 'hotspots' so as to not to overlook dynamic developments in waterbird populations and their environments. Such developments included highly eutrophic lakes subject to rapid overgrowth and urban wetlands.

### 3.2. Sampling geography

Counts were performed on 78 sites (Fig. 1) distributed over 19 of the 43 administrative districts. Some of these points included compact groups of small wetlands, each of which was evaluated as a separate

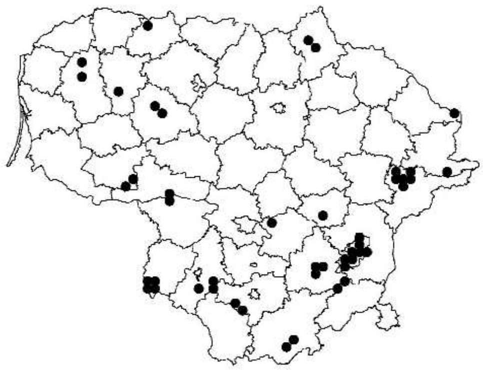

Fig. 1. Network of breeding waterbird monitoring sites in Lithuania (Although only 46 of 78 sites are indicated, eight of the mapped sites include all the remaining 32 sites, which are wetlands, but these comprise aggregations of smaller waterbodies).

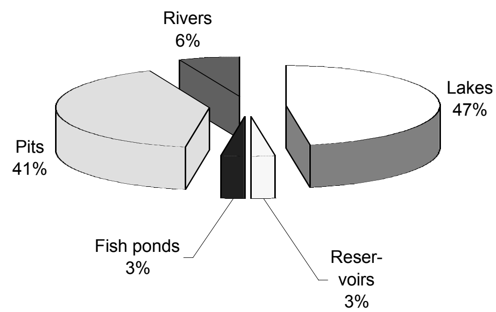


Fig. 2. Numbers of different wetland types in the breeding waterbirds monitoring sample (n=78) in 1999.

point, despite not being distinguished as such within the site records. For example separate points in the city of Vilnius each designate groups of several clay pits.

The uneven distribution of the monitoring sites was the inevitable consequence of the location (and geography) of the protected sites where the skilled observers normally worked, because to minimize the programme costs, the sites were selected predominantly within or in the vicinity of their normal work areas. Additionally, the monitoring network scheme was shaped to some extent by the distribution of lakes in Lithuania. The consequence was that there was a higher concentration of counts along the geographical perimeter of the country than in the central part (see Fig. 1).

### 3.3. Classification of monitoring sites

From the outset, all wetlands in the monitoring sample were classified into lakes and their components (bays, arms and backwaters), water reservoirs and large ponds >1.5ha, , fishponds and small ponds <1.5ha, banks and pits , and river sections. Figs. 2 and 3 illustrate roughly the ecological structure of the monitoring sample, in which the majority of sites contain lakes and small wetlands (*e.g.* pits, sand, clay and peat banks and small ponds) and most of the area of the sites comprises open water- (*e.g.* lakes and fish ponds). The sample thus represents rather well the real situation in Lithuania concerning wetlands. The sample's omission of some streams can be regarded as a shortcoming.

It should be stressed that the grouping of lakes is characterized by great internal diversity, which covers small acid mire lakes located in moss bogs, lakes which have formed where underground gypsum layers have dissolved, small mesotrophic forest lakes and large mesotrophic and

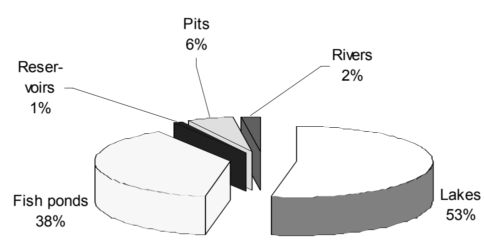


Fig. 3. Proportion of different wetland types in the total area of 3595.6ha covered by monitoring counts in 1999.

eutrophic lakes. Furthermore, the trophic level of the latter types varies within wide limits. A group may comprise large, moderately eutrophic lakes, highly eutrophic lakes and hyper-eutrophic lakes with features of distrophy. Our classification of lakes accords with Kavaliauskiene (1996). The limnological parameters of the monitoring sites will be evaluated in the near future.

### 3.4. Avifauna of Monitored Sites

The results from first year of the programme cannot be used to evaluate the status and perspectives of Lithuanian waterbird populations. However, they show what we can expect to obtain in future. A total of 29 species of waterbirds was counted in 1999 on the 78 wetlands that comprise the monitoring site network. Mallard Anas platyrhynchos, Great Crested Grebe and Coot rank as the most abundant species, followed by Tufted Duck *Aythya fuligula* and Goldeneye *Bucephala clangula* (Fig. 4). Black-headed Gull *Larus ridibundus* deserves special evaluation because of its large breeding concentrations on very few sites. On the other hand, the large numbers in the sample of Garganey *Anas querquedula*, a species with dominant negative trends over most of its breeding range (Farago & Zomerdijk 1997) may be one of those of first useful and unexpected discoveries provided by monitoring.

Mallard, Goldeneye, Coot and Great Crested Grebe, the most abundant species (Fig. 4) also appear to be the most widely distributed (Fig. 5). Only Tufted Duck had narrower breeding habitat requirements than the above species, an d narrower too than Mute Swan *Cygnus olor* and also Garganey, Teal *Anas crecca* and Goosander *Mergus merganser*. These constraints can be demonstrated in the Tufted Duck's avoidance of small shallow wetlands, which are abundant in the sample. In contrast, the Mute Swan tends to occupy such habitats. Although Mute Swan is by no means abundant in comparison with smaller species, because it is territorial

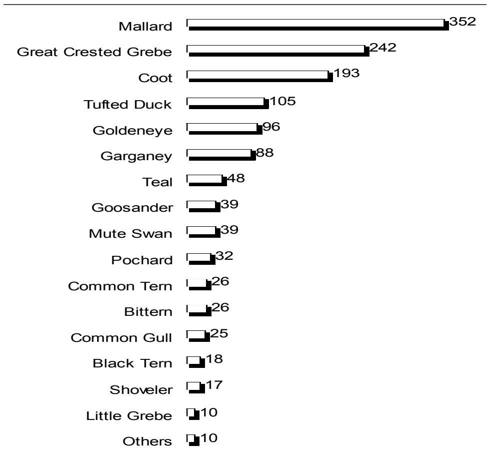


Fig. 4. Number of pairs of 16 (out of 29) breeding waterbird species counted on 78 monitoring sites in 1999. The list does not include Black-headed Gull whose 2224 nests were found in only 3 sites.

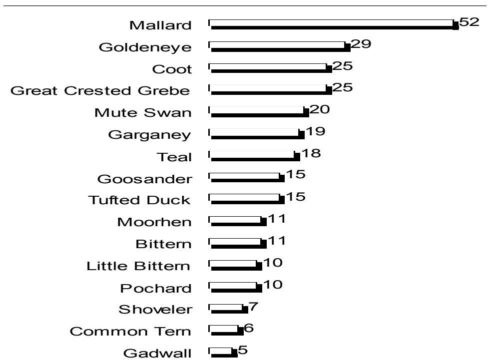


Fig. 5. Occurrence of 16 (out of 29) breeding water bird species on the wetlands monitored. Columns indicate numbers of wetlands in which particular species bred in 1999.

and avoids its congeners, it is able to disperse widely to obtain a breeding site. As expected, the colonial Black-headed Gull was not widely distributed throughout the monitored wetlands. Garganey and Teal attain high rank in the variety of wetlands occupied, that of Teal being explained by its comparatively dominant share of wetlands located in forested landscapes, thesame being true for Goldeneye.

Five other species that were observed frequently on monitoring sites, namely Grey Heron *Ardea cinerea*, Cormorant, Little Gull *Larus minutus*, Herring Gull *L. argentatus* and White-winged Tern *Chlidonias leucopterus* are likely to be found breeding there in the future.

In retrospect, we can say that the results of the first year's work proved that the network of monitoring sites appeared to be quite representative, despite the constraints placed on our management of the programme and on the extent to which we could choose sites randomly. The most abundant and widely-distributed species in the monitoring sample have the same status on country-wide scale (Žalakevičius *et al.* 1995).

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**Ornis Hungarica 12-13: 95-103. 2003**

# Lithuanian wetlands database: the tool for bird monitoring and conservation in Lithuanian wetlands

S. Švažas, L. Balčiauskas and L. Raudonikis

Švažas S., Balčiauskas, L. and Raudonikis, L. 2003. Lithuanian wetlands database: the tool for bird monitoring and conservation in Lithuanian wetlands. – Ornis Hung. 12-13: 95-103.

A countrywide inventory of important wetlands was constructed in Lithuania in 1996-1999. Over 80 wetlands (total area *c*130 000 ha) were investigated. The Ramsar methodology was used as the basis of the inventory's structure. Results of the inventory have revealed that more than 30 Lithuanian wetlands meet the Ramsar criteria qualifying them as of international importance. The wetlands investigated during this survey support the majority of many species of Lithuanian breeding birds, including many that are rare and endangered. Internationally important concentrations of numerous bird species have been recorded in these sites. A high priority of the program was to provide to the regional and local authorities responsible for nature management all necessary information concerning the key wetlands and their birds. Consequently, a new Lithuanian Wetlands Database (WDB) had to be created. It includes the following:
1. Data on important wetlands in each region or district.
2. Data on species and communities in each important wetland.
3. Land use and conservation recommendations at the species or community level.
4. Data visualisation (maps at several levels: from national to local and down to individual wetland).
Replicates of the WDB were forwarded to the organizations responsible for conservation of wetlands in all districts/regions. The WDB permits responses to be made to all matters arising related to wetlands. It will serve as an important tool for further monitoring of bird populations in Lithuanian wetlands and for the formation of new databases of important transfrontier wetlands (those shared by several countries).

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## 1. Introduction

Lithuania contains rich wetland resources. Mires and bogs of all types cover about 5% of the whole country. Large mire complexes comprise almost virgin natural areas that have never been used for agriculture or forestry, the kind of natural habitat that was many years ago in Western Europe. In Lithuania there are 2834 lakes larger than 0.5 ha in area. There are 400 man-made wetlands larger than 5 ha and over 10 000 smaller ponds and reservoirs. Among 758 rivers (longer than 10 km) and numerous streams there are many unregulated or moderately modified rivers with naturally flooded land. The total length of the unregulated natural rivers and streams is *c*17 000 km. The preservation of areas of such extraordinary biological richness is particularly important for many breeding species of birds. In particular, the coastal wetlands (part of the Eastern Atlantic Flyway) are extremely important sites for migratory and wintering populations of waterfowl.

Many important wetlands in Lithuania are at least partly protected, the total protected area now comprising *c*12% of the country. Most preserved areas were originally established when all the land belonged to the State. The recent political and economic changes in Lithuania have during recent years resulted in intensified forestry and in privatisation or re-privatisation of land. Numerous existing wetlands may be turned into private property. In such circumstances it was necessary to compile rapidly the inventory of important wetlands, particularly those areas meeting the criteria of the Ramsar convention. Lithuania joined the Ramsar convention in 1993, 5 key wetlands being designated as Ramsar sites: the Čepkeliai, Kamanos, Viešvile and Žuvintas Strict Nature Reserves and the Nemunas River Delta Regional Park (overall total area *c*50 000 ha). However, most key wetlands lacked detailed habitat and biological community information. The first country-wide inventory of all the important wetlands was compiled in 1996-1999 (Balčiauskas & Švažas 1998), the project being supported by Migratory Birds of Western Palearctic (OMPO). The project objectives were:

- To perform the inventory of the key wetlands, using the Ramsar criteria and classification.
- To produce detailed maps of each important wetland, by plotting important elements of each site.
- To develop the WDB to provide a relevant tool for regional and local decision-makers responsible for wetlands management and to form the basis of further monitoring of wetlands and their fauna and flora.

## 2. Study Area and Methods

More than 80 wetlands (total area *c*130 000 ha) were investigated in 1996-1999. The designated wetlands are comprised of the following wetland types:

1. Mire complexes.
2. Bogs and marshes of all types.
3. Peat-lands.
4. Wet forests.
5. Large shallow lakes.
6. Stretches of river possessing natural floodplain.
7. Natural wet meadows and swamps.
8. Coastal wetlands adjacent to seasonally flooded meadows.
9. Large fishponds and man-made reservoirs.

Intensive field surveys (land-, aerial- and boat-based) were performed in all selected wetlands. The Ramsar datasheet and methodology were used as the basis of the inventory structure. The Ramsar criteria were used to designate wetlands of the international importance. All valuable wetland elements were recorded and plotted on maps. A principal objective of this inventory was to survey breeding and migratory populations of birds.

The Lithuanian WDB was intended primarily as a tool for regional and local decision-makers and to form the basis of further monitoring of important wetlands, their fauna and flora. It is designed to handle spatially oriented data. The data comprises information on rare and common species (although the main emphasis is on birds); the WDB's ability to cope with taxonomic arrangement enables even newly described species to be added. The WDB is compatible with the Lithuanian Biodiversity Database, created by the project group 'Ecological diversity of Lithuania' (Balčiauskas & Budrys 1997) and it includes the species composition and distribution of certain wetlands or districts. Much additional information on species, wetlands and other localities is also contained in the WDB. The WDB was designed under DBMS Paradox for DOS, and includes the mapping module DMAP for DOS (by Alan J. Morton, U.K.).

## 3. Results and Discussion

The results of field surveys performed in 1996-1999 have identified more than 30 Lithuanian wetlands that meet the Ramsar criteria applicable to wetlands of international importance (Švažas *et al.* 1999). Wetlands investigated during this inventory support the majority of many Lithuanian breeding bird species. These wetland areas hold between 70-100% of the total estimated Lithuanian breeding population (see Kurlavičius & Raudonikis 1999) of Black-throated Diver *Gavia arctica*, Greylag Goose *Anser anser*, Shelduck *Tadorna tadorna*, White-tailed Eagle *Haliaeetus albicilla*, Short-toed Eagle *Circaetus gallicus*, Avocet *Recurvirostra avosetta*, Golden Plover *Pluvialis apricaria*, Dunlin *Calidris alpina*, Ruff *Philomachus pugnax*, Great Snipe *Gallinago media*, Black-tailed Godwit *Limosa limosa*, Wood Sandpiper *Tringa glareola*, Short-eared Owl *Asio flammeus*, Aquatic Warbler *Acrocephalus paludicola*, Bearded Tit *Panurus biarmicus* and Great Grey Shrike *Lanius excubitor*. They also support more than 50% of the total Lithuanian population of Black-necked Grebe *Podiceps nigricollis*, Gadwall *Anas strepera*, Pintail *Anas acuta*, Shoveler *Anas clypeata*, Ferruginous Duck *Aythya nyroca*, Red-breasted Merganser *Mergus serrator*, Crane *Grus grus*), Little Tern *Sterna albifrons* and Eagle Owl *Bubo bubo* (Tab. 1). The Nemunas River Delta Regional Park is the most important large-scale nesting centre of waterbird species in Lithuania (Švažas *et al.* 1999, Raudonikis & Kurlavičius 2000). This territory supports the majority of the Lithuanian breeding population of Greylag Goose, Shoveler, Dunlin, Ruff, Black-tailed Godwit, Great Snipe and Little Gull *Larus minutus*. It is the only breeding site of Avocet in Lithuania.

The available data on the population trends of certain species indicate a decline in numbers of Gadwall, Shoveler, Ferruginous Duck, Dunlin, Ruff, Curlew *Numenius arquata* and Black-tailed Godwit in the wetlands covered during this survey. In 1996-1999 these species were not recorded in numerous former breeding grounds. The decline of breeding populations was caused primarily by the increased eutrophication and encroachment of the most important habitats (Žalakevičius *et al.* 1995). The lack of suitable breeding habitats is the main reason of the recent decline of breeding populations of Little Tern and Aquatic Warbler. A rapid increase in numbers of breeding Greylag Geese was recorded in many sites. During the last 20 years this species has become re-established as a breeding species in Lithuania due both to a partial introduction and to a natural influx of birds from adjacent breeding grounds. A high increase in numbers is characteristic of Goosander (*Mergus merganser*), which has recorded in the last decade a rapid range expansion in Lithuania and the other countries of the Eastern Baltic. During the same period, Avocet has also expanded its breeding range, recently forming a new breeding site in the Nemunas river delta area. Small increases in the breeding populations of Bittern *Botaurus stellaris*, Honey Buzzard *Pernis apivorus*, Montagu's Harrier *Circus pygargus*, White-tailed Eagle, Crane and Short-eared Owl were recorded in most wetlands covered in 1996-1999, while the population trends of other analysed species during the last decade have been stable.

The designated and potential Ramsar sites are particularly valuable breeding grounds for many rare, vulnerable and endangered bird species. 40 bird species included in the Lithuanian Red Data Book breed or possibly breed in the Nemunas river delta area and 30 species in the Čepkeliai mire complex. Both wetlands support the largest number of endangered and vulnerable bird species in Lithuania. Several other key wetlands are also distinguished by a very high diversity of protected bird species. Five bird species included in the IUCN Red List of Threatened Animals (Lesser White-fronted Goose *Anser erythropus*, Ferruginous Duck, Steller's Eider *Polysticta stelleri*, Spotted Eagle *Aquila clanga* and Aquatic Warbler) were recorded in 14 wetlands covered during this survey. The main breeding area of Ferruginous Duck is located in southwest and central Lithuania (Lakes Obelija, Metelys, Žuvintas and Žaltytis and the Kauno Marios reservoir). The great majority of Aquatic Warblers breed in wet meadows bordering the eastern coast of the Curonian Lagoon. Up to 200 breeding territories of this species have been recorded in this site during recent years, though high annual fluctuations in numbers of breeding birds are characteristic of this species. Floodplains surrounding the Nemunas river delta are the key stopover sites for Lesser White-fronted Goose in Lithuania. Only single migratory Spotted Eagles in recent years have been recorded in the Nemunas river delta area and in a few inland sites. The marine Palanga site is the only key wintering area of Steller's Eider in Lithuania, up to 2300 birds being recorded. Breeding territories of Corncrake *Crex crex* were recorded in many wetlands investigated in 1996-1999.

The results of this survey have confirmed the huge importance of Lithuanian coastal wetlands for migratory and wintering populations of waterfowl. Internationally important concentrations (exceeding the 1% Ramsar threshold) of 24 waterbird species have been recorded in these sites (Švažas 1996, Švažas *et al.* 1998). The Nemunas river delta area is among the most important staging sites for migratory populations of Whooper Swan *Cygnus cygnus*, Bewick's Swan *Cygnus columbianus bewickii*, White-fronted Goose *Anser albifrons* and Pochard *Aythya ferina* in Europe (Švažas *et al*. 1997, 1998). Large concentrations of Cormorant *Phalacrocorax carbo*, Bewick's Swan, Whooper Swan, Greylag Goose, White-fronted Goose, Bean Goose *Anser fabalis*, Pintail, Goldeneye *Bucephala clangula*, Goosander and Little Gull, exceeding the

1% Ramsar threshold, have been recorded in the northern part of the Curonian lagoon and in the adjacent wet meadows. This brackish lagoon is also the key wintering resort of the wintering population of Goosander, supporting up to 15% of the whole NW Europe population (Švažas *et al*. 1994, Scott & Rose 1996). The shallow inshore marine waters at the coast of Palanga town are among the most important wintering areas of the globally threatened population of Steller's Eider in Europe (Nygard *et al*. 1995). Marine waters along the coast of the Curonian Spit regularly support important concentrations of divers, Great Crested Grebe *Podiceps cristatus*, Velvet Scoter *Melanitta fusca* and Long-tailed Duck *Clangula hyemalis* (Švažas 1993, Vaitkus 1999). Several Lithuanian inland wetlands hold internationally important staging concentrations of waterbirds. The most important are the large shallow lakes of southern and south-west Lithuania. They are the principal staging sites of Great Crested Grebe and Coot *Fulica atra* in Lithuania (Stanevičius 1999). These wetlands regularly support staging flocks of Bean Goose and Crane exceeding the Ramsar 1% threshold. Certain large fishpond complexes are important staging areas of White-fronted Goose and Bean Goose in eastern Lithuania. Furthermore, large peat-lands regularly support important concentrations of Crane (up to 2-4% of the whole NW Europe population) (Raudonikis & Kurlavičius 2000).

All data collected and analysed during this countrywide survey (including all the characteristics of important wetlands in each region or district, distributions of rare species of fauna and flora, recommendations including land use and conservation measures, maps of all levels) were compiled and mapped in the Wetlands Database. Replicates of the newly established database have been distributed to the organisations responsible for the conservation of wetlands in many districts and regions; the WDB will serve as an important tool for wetland management and protection.

The general scope of the WDB is:

1. Accumulation of structured information on wetlands at a national level.
2. Maintenance of regional and district databases.
3. Provision a management tool and information system for local decision-makers.

The WDB comprises several structural parts:

1. Data concerning species and communities, including their distribution and abundance.
2. Data on wetlands.
3. Additional information on species, communities, habitats and wetlands (including their main threats, the status of certain species in Lithuania and in other countries and data on species biology and ecology.).
4. A compendium of special recommendations concerning management and conservation of wetlands, sites, or species.
5. A compendium of general recommendations at the species or community level, including land use practices and management and conservation measures.
6. Data visualisation (mapping and plotting at several levels, from the national to local and down to selected wetlands).

A simplified relational structure of the

WDB and how its elements are interconnected is provided in Fig. 1. The end-user interface is made as simple as possible to allow WDB users in all Lithuanian municipalities and districts to use it without special training. The WDB provides the possibility of managing data without having to depend on a mass of instructional tables, forms or reports. The end-user interface consists of menus and dialogue boxes.

The WDB lacks GIS-functionality, because it is not a tool for land-use planning. However, the Lithuanian Ministry of Environment will establish special GIS systems for this purpose. The WDB includes several levels of spatial representation, from national down to local level (Figs. 2-5). Starting at national level, the WDB represents a map of all wetlands included into the database (Fig. 2a); it also shows the location of the selected wetland (Fig. 2b) and displays species distribution with dots that indicate the sites with species records (Fig. 2c):

At the administrative district level, the WDB represents a map of the selected district showing the location of each important wetland in the district (Fig. 3a), the location of the selected wetland (Fig. 3b) and sites with records of rare species of fauna and flora in district wetlands (Fig. 3c).

At a wetland (local) level the territorial representation is restricted to a schematic diagram of the selected wetland (including boundaries of the area, district or state borders, main roads, position of the nearest settlements, lakes, rivers and other water bodies) (Fig. 4a). The same infor-

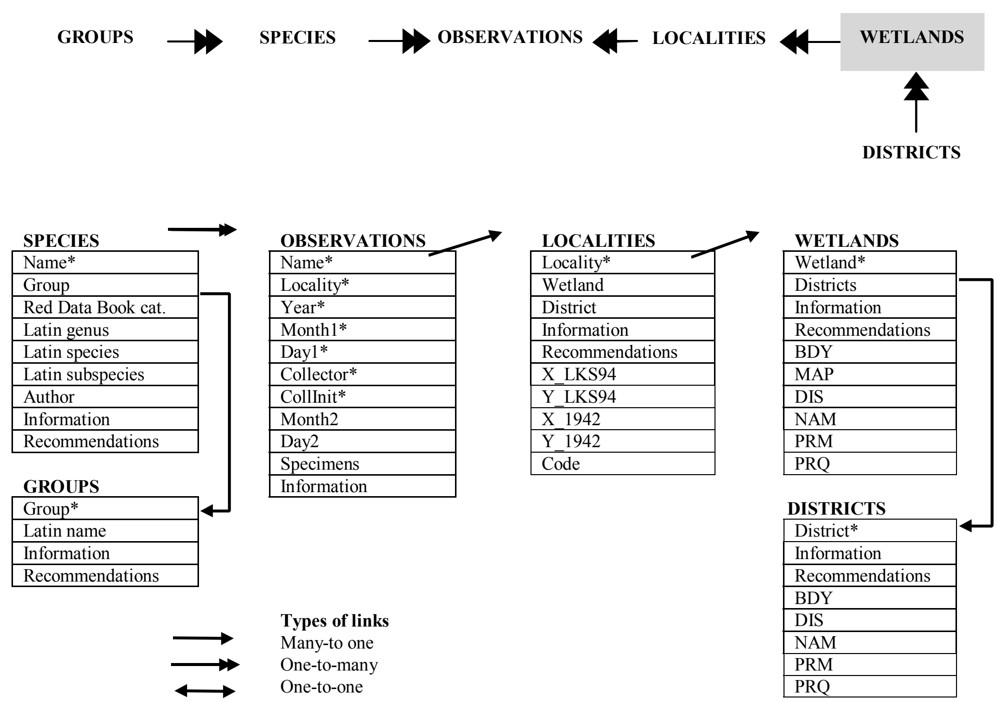


Figure 1. The structure of data files of the Wetland Database. The links between data fields of these files are shown as arrows.


Figure 2. The Territorial output of the WDB at a national level, representing: a. The map of important Lithuanian wetlands. b. Location of the selected wetland (the Nemunas River Delta). c. Distribution of one species in Lithuania.

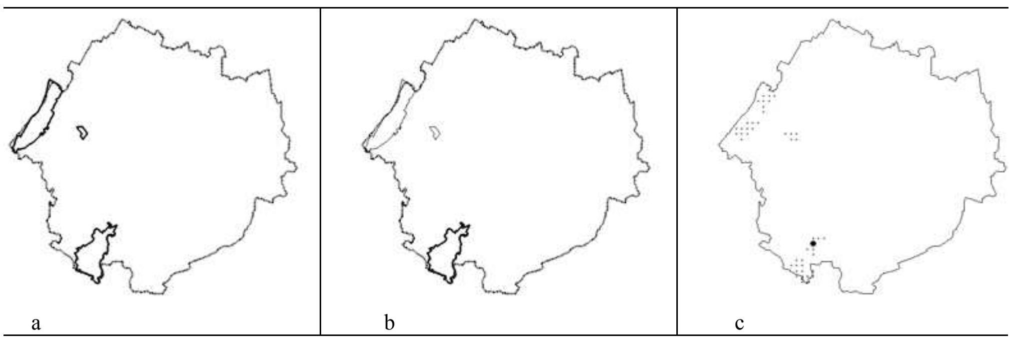


Figure 3. Territorial output of the WDB at a district level, representing: a. Location of all the important wetlands in one administrative district. b. Location of the selected wetland in the district. c. Localities of all survey points in important wetlands of the district, the selected point being indicated as black dot.

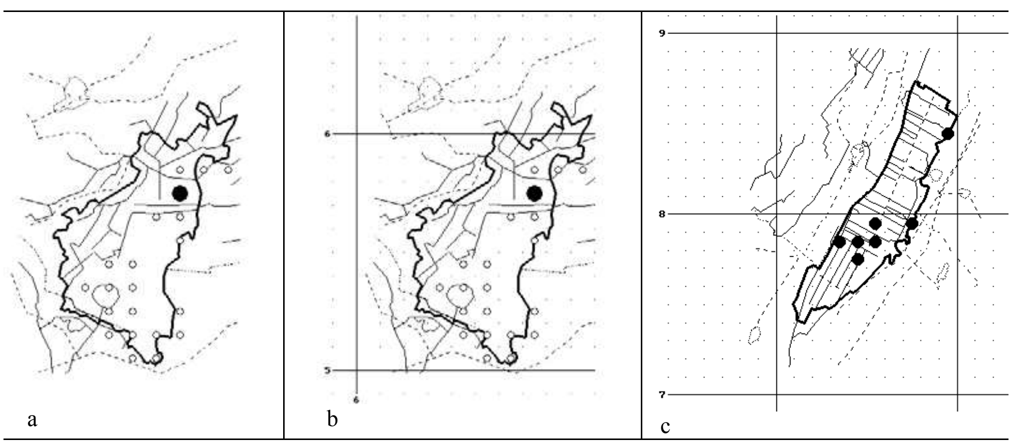


Figure 4. Territorial output of the WDB at a local or wetland level, representing: a. Location of all survey points in the wetland (unfilled, or empty dots), location of the selected locality (solid, or black dot), boundaries of the area (thick solid line), district or state borders (thin dot-dash line), main roads (thin dash line), nearest settlements (thin dot line), water bodies (thin solid line). b. The same output using the 10x10km grid coordinate system (lines) and the 1×1 km grid (dots). c. Distribution of one selected species in a particular wetland.

mation can be presented with coordinate gridlines representing 10×10 km squares and dots 1×1 km squares (Fig. 4b). At this level, the position of survey points (or localities) is represented on the scheme by unfilled dots; the position of the selected locality is emphasized as a solid (black) dot. On the schematic diagram of species distribution at a wetland level, solid (black) dots represent localities where the species has been recorded (Fig. 4c). This information is mapped directly from the database tables.

The territorial approach in the Wetland Database is realised according to the following scheme: District→Wetland→ Locality→Observation. There are several possibilities for the end-user at every link of this chain, including:

1. General information; *e.g.* the size of wetlands in the district, the habitat structure of the selected wetland, species composition in selected sites.
2. Recommendations concerning wetlands management and conservation.
3. Input of new data or information.
4. Editing the existing data and information.
5. The list of the elements involved; *e.g.* the list of species in a selected wetland.
6. Context-specific help system.
7. The map output, implemented in the first two levels.

The species approach is formed according to the following scheme: Species Group→Species. At the species level there are several choices: 'species in districts', 'species in wetlands' and 'species in certain sites'. It is possible to work only with species included in the Lithuanian Red Data Book. The possibilities for the end-user at every link of this chain are almost the same as in the previous scheme. The lowest level of information in the WDB are observations, including both territorial and species approaches. The observation is defined as a unique combination of place, species, time and observer. Starting from the observation, the biodiversity estimation is performed in the same way as at other levels, species lists being annotated with locality, wetland and district information. Estimation is calculated as the Hill's diversity numbers N1, N2 and N3, Shannon's diversity indices with base of *e* or 2, and Simpson's evenness index (see Ludwig & Reynolds 1988).

It is expected that through use of the WDB, results from the wetlands inventory program will provide direction and support for the most effective and forthcoming practical implementation of recommendations concerning the protection and subsequent management of important Lithuanian wetlands during the period of national economic transition. The newly created WDB can form the structural basis of further monitoring of Lithuanian wetlands and their fauna and flora. At present it includes all the available data on the essential elements of important wetlands (particularly on birds), collected during field surveys performed in 1996-1999 and compiled from the published sources. The data in the WDB on species, communities, wetlands and the resultant recommendations may in future be supplemented, edited or deleted, according to subsequent studies. All future data on the included wetlands or species will be incorporated into the WDB. Researchers from universities or research institutes, the staff of National and Regional Parks and municipal ecologists will add new data into databases at district or local level for transfer into the national WDB functioning in the Institute of Ecology.

The Lithuanian Wetlands Database will be also used for the formation of a new Regional Database of Important Transfrontier Wetlands. The inventory of important wetlands shared by Lithuania with Belarus and with Russia will be implemented in 2001-2002. All data collected during this international program will be compiled and mapped in a special database whose structure is based on the WDB.

*Acknowledgments*. The inventory of important wetlands in Lithuania was implemented through the enthusiastic participation of a large team of ornithologists, ecologists, foresters and amateur naturalists. This program would not have been possible without the support of OMPO ('Migratory Birds of the Western Palearctic'). Dr. E. Budrys has contributed significantly to the design and implementation of the Wetlands Database.

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**Ornis Hungarica 12-13: 105-113. 2003**

# A low-cost, year-round seabird monitoring programme in the English channel and Bay of Biscay: preliminary results 1995-2001

T. M. Brereton, C. Weir, M. Hobbs and A. D. Williams

Brereton, T. M., Weir, C., Hobbs, M. and Williams, A. D. 2003. A low-cost, year-round seabird monitoring programme in the English channel and Bay of Biscay: preliminary results 1995-2001. – Ornis Hung. 12-13: 105-113.

In 1995, the Biscay Dolphin Research Programme established a monthly, year-round seabird and cetacean monitoring programme in the western English Channel and eastern Bay of Biscay, using the P&O Portsmouth cruise-ferry the Pride of Bilbao and volunteer surveyors. On each four-day survey trip bird recording was made during all daylight hours, enabling the whole of the route to be sampled over much of the year. Over the 5.25 year recording period 44 seabird species were recorded, with more than 100 000 seabirds counted in approximately 50 000 km of search effort, spread over 85 ICES rectangles (measuring 15' latitude by 30' longitude). Shelf waters of the northern Bay of Biscay were found to have presumed important numbers of wintering Northern Gannet *Morus bassanus* and Great Skua *Catharacta skua* at relatively high densities. A number of other species including Divers *Gaviiformes*, gulls *Laridae* and auks *Alcidae* were present at lower abundance. During the summer, these shallow waters supported moulting populations of Mediterranean Shearwater *Puffinus yelkouan (mauretanicus)* at the northern edge of their range and European Storm-petrels *Hydrobates pelagicus*, which are scarce in north European waters at this time of year. A number of seabirds rare elsewhere in north European waters were regularly recorded during the late summer and autumn period in shelf-edge and deep water areas of the Bay of Biscay, including Little Shearwater *Puffinus assimilis*, Cory's Shearwater *Calonectris diomedea*, Great Shearwater *Puffinus gravis*, Grey Phalarope *Phalaropus fulicaria*, and Sabine's Gull *Larus sabini*. Spectacular numbers of Great and Cory's Shearwaters were found in most years of survey, with loafing flocks of hundreds of birds seen on numerous occasions. Recent methodology enhancements have been made to improve the quality of monitoring data, and further improvements are proposed. The BDRP surveys have demonstrated how volunteers and ferries can be used to generate low-cost, quantitative data in areas where large -scale systematic surveys are unlikely to be carried out. The results of such surveys may have considerable conservation and policy relevant implications, particularly in the designation and year-round management of marine Important Bird Areas.

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## 1. Introduction

This paper presents baseline monitoring data on the distribution and relative abundance of seabirds in offshore waters of the eastern Bay of Biscay and western English Channel. These data derive from a preliminary analysis of sightings data collected through Biscay Dolphin Research

Programme (BDRP) surveys, between 1995 and 2001. BDRP is a voluntary marine research, education and conservation organisation, sponsored by P&O Portsmouth and affiliated to Sea Watch Foundation.

There have been relatively few offshore seabird surveys in the eastern Bay of Biscay and the Western Approaches of the English Channel (White & Reid 1997) in comparison with surrounding UK waters and the eastern North Sea. The European Seabirds at Sea Team survey/database area extends as far south as the north coast of Brittany, but not into the Bay of Biscay (Webb pers comm). Seabirds at Sea (SAS) surveys in the western English Channel have located internationally important populations of Great Skua *Catharacta skua*, Northern Gannet *Morus bassanus* and Little Gull *Larus minutus* (Stone *et al*. 1995).

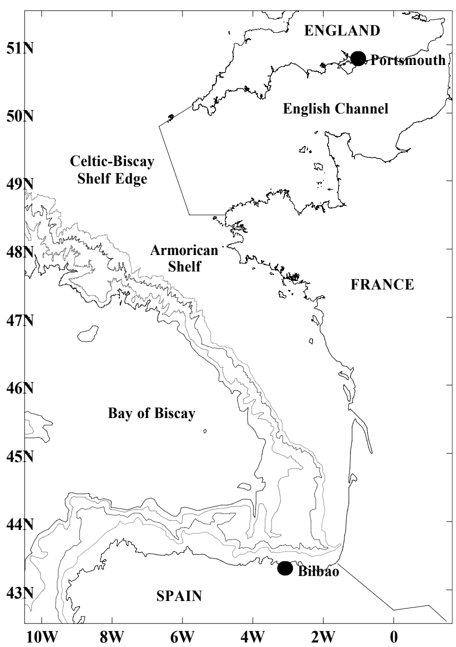
Seabird survey and distribution data from the Bay of Biscay are patchy, available published data mainly describing individual species. Hémery & Jouanin (1988) analysed survey and other data of Leach's Petrels *Oceanodroma leucorhoa*, estimating a Biscay winter population of 0.3-2 million birds. Le Mao & Yésou (1993) estimated the late summer coastal population of Yellow-legged Gulls *Larus cachinnans* along the French coast of the Bay of Biscay at approximately 20 000 birds. Yésou (1986) further estimated that 8-10 000 Mediterranean Shearwaters *Puffinus yelkouan (mauretanicus)* undertake a post-breeding moult in French coastal waters. A recent study by Guerin (1999) has identified a similar early summer coastal occupancy of Cory's Shearwater *Calonectris diomedea* off northwest France. Offshore seabird surveys in the eastern Bay of Biscay have been undertaken and compiled by Bourne (1986). More recently transect surveys off the northwest coast of France have been carried out by local French groups (Webb pers comm.). Casual records from bird watchers have been collated periodically from the Bay of Biscay. In 1990, Seawatching and Birding Alternatives produced a report of sightings from Plymouth (England) to Santander (Spain) crossings. Since 1997 England-Spain ferry sightings have been collated by Organisation Cetacea and summarised in annual reports.

Until recently, marine environments have received little attention in terms of identifying Important Bird Areas (IBAs) due to the difficulty in defining and delineating individual sites (Heath & Evans 2000). Skov *et al*. (1995) identified the coastal waters off Start Point (Devon, southern England) in the western English Channel as an IBA. Not enough data are available to assess accurately IBA status in other parts of the Channel (Skov *et al*. 1995) and no work has been done to identify candidate IBAs in offshore waters of the Bay of Biscay (Heath & Evans 2000).

In 1995, BDRP launched a year-round seabird-monitoring programme in the western English Channel and eastern Bay of Biscay, the primary aim being to generate baseline data on the seasonal distribution and abundance of seabirds in these under-sampled regions.

## 2. Study area and methods

The study area is situated in temperate waters of the northeast Atlantic, between



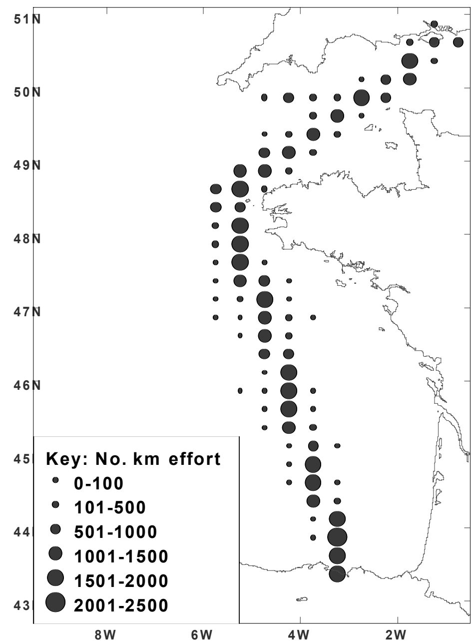




Fig. 1. (a) Study area and (b) survey coverage 1995-2001 (In Fig. 1a depth contours from right to left are 200 m, 1000 m, 2000 m, and 4000 m.).

latitudes 43° to 51°N and 0° to 8°W and comprises two major seas, the English Channel and Bay of Biscay (Fig. 1a). For the purposes of this paper, the two seas have been demarcated by latitude, with seabird sightings north 48°30'N classified as within the English Channel, whilst those to the south in the Bay of Biscay.

BDRP seabird survey work in the English Channel and Bay of Biscay was undertaken exclusively from the P&O Portsmouth cruise-ferry, the Pride of Bilbao. The ferry sails between Portsmouth in southern England and Bilbao in northern Spain, traversing British, French and Spanish waters. The ferry follows a set route, but the course alters at times and in total effort-related sightings data have been obtained from 85 ICES (International Council for the Exploration of the Sea) rectangles, measuring 15' latitude by 30' longitude (Fig. 1b). The English Channel consists of shallow continental shelf waters that rarely exceed 100 m in depth. Depth increases slightly to between 100 and 200 m over the Armorican Shelf off the southwest coast of Brittany. The main feature of the survey area is the region of steep continental slope, which marks the border between continental shelf waters and the abyssal plain of the Bay of Biscay. Along the slope, water depth drops from 200 m to over 4000 m over distances as little as 35 km. The Bay of Biscay predominantly is a deep-water area, approaching depths of 5000 m in some places. In the southern portion of the Bay, seabed topography is highly variable and two deep-water canyons extend to within 17 km of the Spanish coastline. The easternmost canyon known as the Cap De Breton Canyon, is traversed during the ferry crossing.

Tab. 1. Survey effort (km travelled) by season in the English Channel and Bay of Biscay on bird recording trips.

| Season (Months) | No of trips | English Channel km travelled | Bay of Biscay km travelled |
|---|---|---|---|
| Spring (Mar-Jun) | 17 | 8683 | 11912 |
| Summer/Autumn (July-Oct) | 18 | 5198 | 9830 |
| Winter (Nov-Feb) | 16 | 7338 | 5629 |
| Total | | 21219 | 27370 |

Year-round, monthly seabird surveys were undertaken from the Pride of Bilbao ferry. Each return crossing extended over four days, allowing the whole route to be sampled at least once during daylight hours in summer and approximately 75% of the route in winter. The main gap in coverage was the northern Celtic-Biscay shelf-edge (45°N-46°30'N).

On each trip, effort-related seabird recording was carried out during all available daylight hours by a team of three experienced volunteer observers. Recording was carried out from a fixed position on the bridge of the ship, at a height of 32 m and speed of 15-22 kt. Each seabird observed ahead of the ship was counted once only, with sightings grouped into minute-long periods. Between 1995 and July 2000, all birds within an assumed 2 km-wide strip either side of ahead were counted. For each seabird sighting the following records were made: species name, number seen, and where possible age and sex. Notes on behaviour were also made, including associations with fishing vessels, cetaceans, and environment effects such as oiling and fishing net entanglement. From August 2000, refinements were made to the methodology, to enable estimation of bird density in ICES rectangles. All birds observed ahead of the ship were recorded once only as either (1) inside an imaginary 300 m recording box on the starboard side, or (2) outside the 300 m box (but presumed within 2 km).

Effort data was collected simultaneously with sightings data, to enable the number of sightings to be scaled to recording effort and (to detect change) the calculation of relative frequency, abundance and density. At 15 to 30-minute intervals, or whenever the ship's course changed, a range of variables was measured, including the ship's speed and course, and those of sea and weather such as sea state and visibility (Evans 1995).

Between August 1995 and February 2001, 63 (four-day) survey trips were made, resulting in more than 65 000 km of completed search effort. Bird data were collected on 51 trips, totalling just under 50 00 km of survey effort, even coverage being attained through the seasons (N=16-18 trips, Tab. 1).

The survey effort travelled, in km and by ICES rectangle (30' latitude by 15' longitude), and the latitude and longitude of all bird sightings (from timings) were calculated by formulas in MS Excel. Distribution maps have been generated by DMAP biological mapping software supplied by Dr Alan Morton.

## 3. Results

Over the 5.25-year recording period 44 seabird species were recorded (Tab. 2), more than 100 000 seabirds being counted in approximately 50 000 km of search effort. Northern Gannet was the most abundant species in both seas, but a full analysis of this species has yet to be completed, due to the volume of records, although a preliminary inspection of the

Tab. 2. Bird species recorded in the eastern Bay of Biscay and western English Channel on BDRP surveys 1995-2001. Ten most abundant species given in ascending rank order.

| SEABIRD FAMILY | SPECIES | BISCAY COUNT (RANK) | CHANNEL COUNT (RANK) | GRAND TOTAL |
|---|---|---|---|---|
| Gaviidae | Red-throated Diver *Gavia stellata* | 0 | 7 | 7 |
| Gaviidae | Black-throated Diver *Gavia arctica* | 0 | 2 | 2 |
| Gaviidae | Great Northern Diver *Gavia immer* | 0 | 7 | 7 |
| Gaviidae | Diver sp | 1 | 3 | 4 |
| Podicipedidae | Black-necked Grebe *Podiceps nigricollis* | 7 | 0 | 7 |
| Procellariidae | Northern Fulmar *Fulmarus glacialis* | 653 (7) | 803 (4) | 1456 |
| Procellariidae | Cory's Shearwater *Calonectris diomedea* | 3308 (5) | 7 | 3315 |
| Procellariidae | Great Shearwater *Puffinus gravis* | 6175 (2) | 4 | 6179 |
| Procellariidae | Sooty Shearwater *Puffinus griseus* | 97 | 21 | 118 |
| Procellariidae | Little Shearwater *Puffinus assimilis* | 10 | 0 | 10 |
| Procellariidae | Manx Shearwater *Puffinus puffinus* | 70 | 91 (10) | 161 |
| Procellariidae | Mediterranean Shearwater *P. yelkouan mauretanicus* | 49 | 8 | 57 |
| Procellariidae | Shearwater sp. | 0 | 1 | 1 |
| Procellariidae | Small Shearwater sp | 1 | 2 | 3 |
| Procellariidae | Cory's/Great Shearwater | 391 | 12 | 403 |
| Hydrobatidae | European Storm-petrel *Hydrobates pelagicus* | 314 (9) | 407 (9) | 721 |
| Hydrobatidae | Leach's Storm-petrel *Oceanodroma leucorhoa* | 3 | 10 | 13 |
| Hydrobatidae | Wilson's Storm-petrel *Oceanites oceanicus* | 3 | 0 | 3 |
| Hydrobatidae | Madeiran Storm-petrel *Oceanodroma castro* | 1 | 0 | 1 |
| Hydrobatidae | Petrel sp | 35 | 12 | 47 |
| Diomedeidae | Black-browed Albatross *Diomedea melanophris* | 1 | 0 | 1 |
| Sulidae | Northern Gannet *Morus bassanus* | 4500**(1) | 6376**(1) | 10876 |
| Phalacrocoracidae | Great Cormorant *Phalacrocorax carbo* | 269 | 52 | 321 |
| Phalacrocoracidae | European Shag *Phalacrocorax aristotelis* | 7 | 3 | 10 |
| Anatidae | Common Scoter *Melanitta nigra* | 50 | 62 | 112 |
| Phalaropodidae | Grey Phalarope *Phalaropus fulicaria* | 21 | 4 | 25 |
| Stercorariidae | Pomarine Skua *Stercorarius pomarinus* | 70 | 14 | 84 |
| Stercorariidae | Arctic Skua *Stercorarius parasiticus* | 40 | 6 | 46 |
| Stercorariidae | Long-tailed Skua *Stercorarius longicaudus* | 5 | 2 | 7 |
| Stercorariidae | Great Skua *Catharacta skua* | 915 (6) | 627 (6) | 1542 |
| Stercorariidae | Arctic/Long-tailed Skua | 1 | 3 | 4 |
| Stercorariidae | Pomarine/Arctic Skua | 10 | 9 | 19 |
| Laridae | Mediterranean Gull *Larus melanocephalus* | 18 | 51 | 69 |
| Laridae | Little Gull *Larus minutus* | 307 (10) | 41 | 348 |
| Laridae | Sabine's Gull *Larus sabini* | 172 | 10 | 182 |
| Laridae | Black-headed Gull *Larus ridibundus* | 505* | 2588* | 3093 |
| Laridae | Common Gull *Larus canus* | 3 | 940* | 943 |
| Laridae | Kittiwake *Rissa tridactyla* | 4631 (4) | 3174 (3) | 7805 |
| Laridae | Lesser Black-backed Gull *Larus fuscus* | 5430 (3) | 3910 (2) | 9340 |
| Laridae | Herring Gull *Larus argentatus* | 166 | 1961* | 2127 |
| Laridae | Yellow-legged Gull *Larus cachinnans* | 2500* | 1 | 2501 |
| Laridae | Great Black-backed Gull *Larus marinus* | 409 (8) | 753 (5) | 1162 |
| Laridae | Gull sp. | 56 | 168 | 224 |
| Laridae | Large Gull sp | 2304 | 1314 | 3618 |
| Laridae | Small Gull sp | 26 | 2 | 28 |
| Sternidae | Sandwich Tern *Sterna sandvicensis* | 39 | 16 | 55 |
| Sternidae | Common Tern *Sterna hirundo* | 69 | 563 (7) | 632 |
| Sternidae | Arctic Tern *Sterna paradisaea* | 22 | 7 | 29 |
| Sternidae | Little Tern *Sterna albifrons* | 2 | 1 | 3 |
| Sternidae | Black Tern *Chlidonias niger* | 5 | 3 | 8 |
| Sternidae | Tern sp | 71 | 2 | 73 |
| Sternidae | Common/Arctic tern | 142 | 30 | 172 |
| Alcidae | Common Guillemot *Uria aalge* | 290 | 418 (8) | 708 |
| Alcidae | Razorbill *Alca torda* | 37 | 63 | 100 |
| Alcidae | Little Auk *Alle alle* | 1 | 0 | 1 |
| Alcidae | Atlantic Puffin *Fratercula arctica* | 54 | 30 | 84 |
| Alcidae | Auk sp | 93 | 146 | 239 |

* Coastal species with incomplete counts (not included in species rankings).
** Northern Gannet data incomplete in table, but top ranking species in both regions.
Categories unidentified to species level were not included in ranking.bles

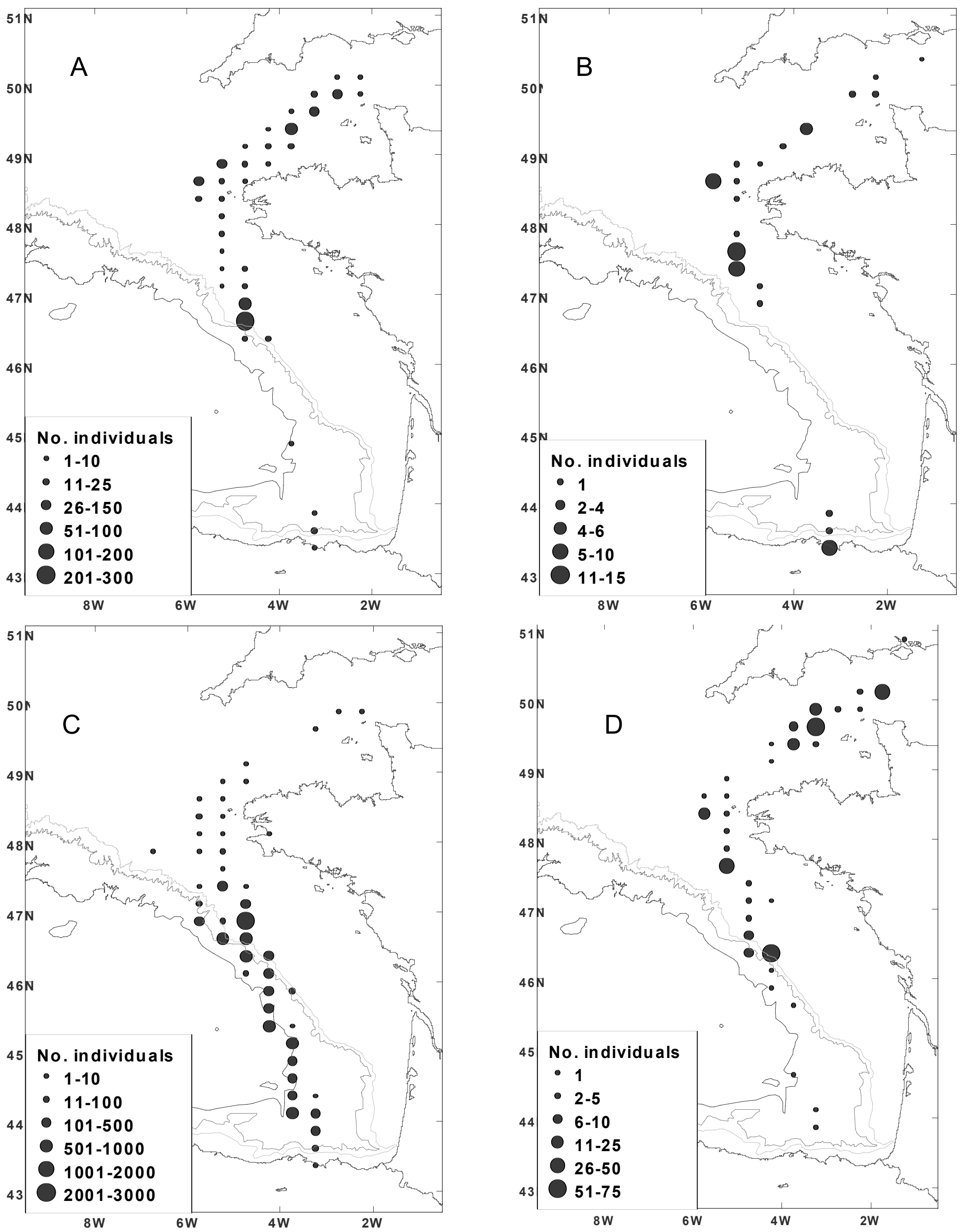



Fig. 2. Distribution and relative abundance of selected seabirds (Depth contours from right to left 200m, 1000m, 4000m). (a) Great Skua November to February, (b) Atlantic Puffin November to February, (c) Great & Cory's Shearwaters Jul-Nov, (d) European Storm-petrel Aug-Oct.

data indicates the largest numbers are found during the late winter period, in the western English Channel and the Armorican Shelf of northern Biscay. This region was also found to support a number of other wintering birds including high

Tab. 3. Bird species which may occur in offshore waters of the western English Channel and eastern Bay of Biscay at levels of regional and north European conservation importance.

| Species | Area | Timing | Likely conservation significance |
|---|---|---|---|
| Cory's Shearwater | Biscay | July-Oct | International |
| Great Shearwater | Biscay | Aug-Nov | International |
| Mediterranean Shearwater | Biscay/W Channel | Jul-Nov | International |
| European Storm-petrel | N Biscay/W Channel | Aug-Oct | Regional |
| Northern Gannet | N Biscay/W Channel | Nov-Feb | International |
| Great Skua | N Biscay/W Channel | Nov-Feb | International |
| Kittiwake | Biscay/W Channel | Nov-Feb | Regional |
| Lesser Black-backed Gull | Biscay/W Channel | All year | Regional |
| Razorbill | Biscay/Channel | All year | Regional |
| Common Guillemot | Biscay/Channel | All year | Regional |
| Atlantic Puffin | Biscay/Channel | All year | Regional |

densities (0.5 birds/km$^2$, Feb 2001) of Great Skua (Fig. 2a) and lower densities of Little Gull, Razorbill *Alca torda*, Common Guillemot *Uria aalge*, Atlantic Puffin *Fratercula arctica* (Fig. 2b) and several gull species. Great Skuas and auks were also recorded regularly in the shelf waters of the southern Bay of Biscay.

Relatively large numbers of Kittiwake Rissa tridactyla and Lesser Black-backed Gull *Larus fuscus* were found in both seas, especially in association with fishing boats. For Lesser Black-backed Gull, a substantial proportion (33%) of sightings were recorded in March and April, presumably indicating a passage of birds through their area from south European wintering grounds. During winter, a high proportion of those aged (e.g.more than 75%, November 2000) recorded off the Brittany Coast were considered to be one of the northern European races *L.f. intermedius* that breeds only in Netherlands, Denmark and southern Norway (Snow & Perrins 1998) and is scarcer than most European gulls at sea. Approximately three-quarters of Kittiwake sightings were recorded during the winter months, confirming the importance of these shelf waters for the species.

A number of seabirds rare in north European waters were regularly recorded during the late summer and autumn period in the Bay of Biscay including Mediterranean Shearwater, Cory's Shearwater, Great Shearwater *Puffinus gravis*, Grey Phalarope *Phalaropus fulicaria* and Sabine's Gull *Larus sabini*. Spectacular numbers of Great and Cory's Shearwaters (Fig. 2c) were found in most years of survey, with loafing flocks of hundreds of birds seen on numerous occasions. Both species were widely distributed from 43°30' to 47°30'N in water depths of 100 m to over 4000 m. Wilson's storm-petrel *Oceanites oceanicus* and Little Shearwater *Puffinus assimilis* were seen on a number of occasions, and are known from casual records to occur with regularity.

Petrels (Hydrobatidae) were undoubtedly under-recorded due to the height of observation and speed of travel. European Storm-petrel *Hydrobates pelagicus* was by far the most frequently recorded petrel, the majority (80%) of sightings occurring between August and October in shelf waters (Fig. 2d). Leach's Petrels *Oceanodroma leucorhoa* were exclusively seen during September and October, with no evidence of a large wintering population as described by Hémery & Jouanin (1988).

## 4. Discussion

There are a number of likely reasons why the outer reaches of the English Channel and Bay of Biscay have been less well surveyed for seabirds, than for example coastal UK waters and the North Sea. The Channel and Biscay have been presumed to hold lower densities of seabirds, especially auks, vulnerable to pollution incidents. Both seas are relatively inaccessible and are notorious for stormy weather, making surveys logistically more difficult. Perhaps the most important reason is that exploitation for oil and gas has been little developed. Dedicated seabird surveys are very costly. The majority of European offshore surveys so far completed have been achieved only in an economic context as part of environmental impact assessments for the oil and gas industries.

For these reasons, funding is unlikely to be available in the near future for large-scale dedicated surveys. Low-cost methods using ferries therefore are likely to be the only realistic option of getting data from these regions. BDRP surveys in the western English Channel and Bay of Biscay have provided a wealth of new data on the distribution and status of seabirds in these waters and will act as a baseline from which to monitor future changes. New information has been generated on the precise range of a number of species including Little Shearwater, Cory's Shearwater, Great Shearwater, Great Skua and Atlantic Puffin.

The ferry route samples all the major topographical features and underwater habitats found in the English Channel and Bay of Biscay, but cannot be considered fully representative of these seas and consequently population estimates cannot be derived from the survey results. However, from a preliminary analysis of the data, it is possible to speculate on the likely species occurring in both seas in numbers of conservation significance (Tab. 3).

Future analyses will concentrate on mapping the seasonal abundance of all species, and generating more density estimates. Throughout the survey period, the recording priority of BDRP has remained to generate reliable monitoring data on cetaceans. Because of a combination of manpower limitations, the ship's extraordinarily high viewing platform and the rapid speed of the ship, it has not been thought possible to record seabirds using standard seabird monitoring methods (Tasker *et al*. 1984). As a consequence, calculated seabird density estimates will not be directly comparable, as correction factors cannot be applied for birds such as auks and petrels, whose detectability decreases with distances from the survey vessel. These species will be particularly under-recorded on the current survey vessel, owing to the viewing height. Subject to resources, future BDRP surveys for auks and petrels will use standard methods. It is thought that the panoramic view provided by the survey vessel should enable accurate densities of visible species such as Northern Gannet, Great Skua, Great Shearwater, Cory's Shearwater and gulls, and hence no change of methodology is planned for these species.

The BDRP surveys have demonstrated how volunteers and ferries can be used to generate low-cost, quantitative data that may have considerable conservation and policy relevant implications, particularly in the designation and year-round management of marine IBAs.

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**Ornis Hungarica 12-13: 115-125. 2003**

# Baseline Monitoring data on Procellariformes (Shearwaters) in the Bay of Biscay

M. J. Hobbs, T. Brereton, C. Weir and A. Williams

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This paper presents baseline data concerning the year round status and distribution of shearwater (Procellariforme) species in the Bay of Biscay and western English Channel from a preliminary analysis of sightings data collected through Biscay Dolphin Research Programme (BDRP) surveys between September 1995 and February 2001. Six species of shearwater were recorded in the Bay of Biscay through BDRP surveys; Cory's Shearwater *Calonectris diomedea*, Great Shearwater *Puffinus gravis*, Sooty Shearwater *Puffinus griseus*, Manx Shearwater *Puffinus puffinus*, Mediterranean Shearwater *Puffinus yelkouan (mauretanicus)*, and Little Shearwater *Puffinus assimilis*. The two large shearwater species are largely present from late July to early October in the deeper waters (>1000m) of the Bay (44-47°N). The largest concentrations of birds are present at presumed areas of upwelling near the northern Celtic-Biscay shelf edge, where flocks of up to 1000 Great and 400 Cory's Shearwater have been recorded from surveys. Unlike Cory's Shearwater, Great Shearwater is often still found in large numbers up to late October. Sooty Shearwater does not occur in large numbers but is distributed throughout the survey area on migration. Manx Shearwater is found in the largest numbers in the English Channel and off the Brittany coast in the April-June period and is largely absent from the Bay itself. Mediterranean Shearwater occurs in low numbers with a fairly even distribution of records from May to November. Little Shearwater is a rare although regular species August-October.

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## 1. Introduction

The Bay of Biscay has long been known as an important stopover for various shearwater species, although there is comparatively little evidence of systematic seabird surveys within the area, in comparison to UK waters and the North Sea. Great Shearwater has been recorded in high densities along the northern Celtic/Biscay shelf edge (Bourne 1986, Voous & Wattel 1963). Cory's Shearwater is known to disperse into the Bay in the summer months; non-breeders from May-June and other birds, in large numbers from July-October (Cramp 1977, Guerin 1999). The Bay is also known as an important moulting site for Mediterranean Shearwaters of the race *P. y. mauretanicus*, which congregate in coastal areas on the shelf (Le Mao & Yésou 1993). There is little historical evidence that the area is important for Manx, Sooty or Little Shearwater.

The Biscay Dolphin Research Programme is a voluntary marine research and conservation organisation, established in 1995 and sponsored by P&O Portsmouth. The aim of the first five years of study was to establish baseline data on

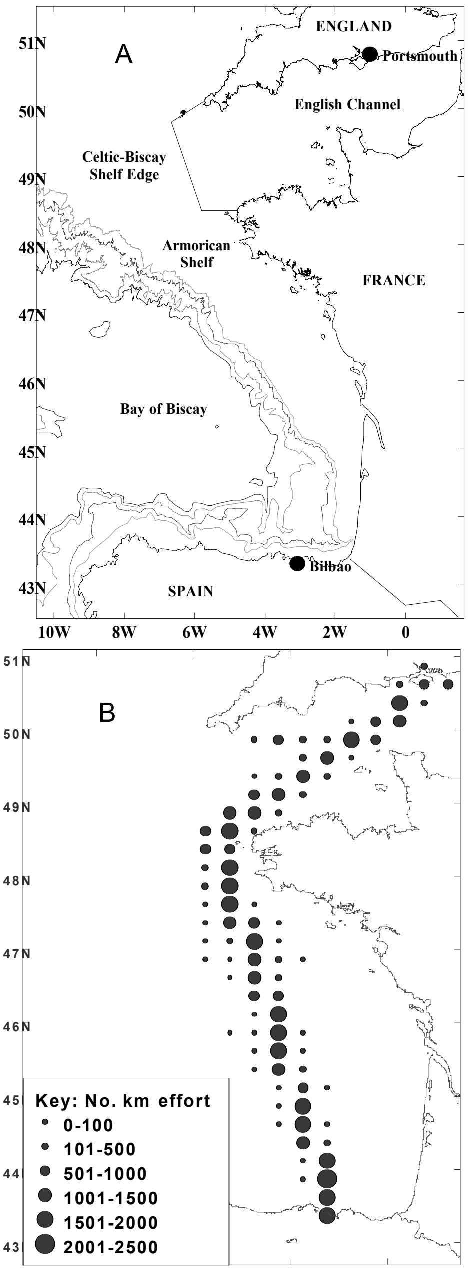


Fig. 1. Study area and survey coverage 1995-2001, (a) Study area, (b) Survey coverage (1a: depth contours from right to left are 200m, 1000m, 2000m, 4000m).

the distribution, behaviour and relative abundance of cetaceans, seabirds and other marine organisms along the route of the Pride of Bilbao, a commercial ferry which sails from Portsmouth (England) to Bilbao (northern Spain).

## 2. Study area and methods

The study area is situated in temperate waters of the northeast Atlantic, between latitudes 43°N to 51°N and 0°W to 8°W. It comprises two major sea areas, the English Channel and Bay of Biscay (Fig. 1a).

Throughout the survey period, the recording priority of the BDRP was to generate reliable monitoring data on cetaceans. Seabird recording necessarily came second, and this in combination with the limited manpower, the rapid speed of the ship and the extraordinarily high viewing platform, militated against recording seabirds using standardised seabird monitoring techniques (Tasker *et al*. 1984). Seabird surveys were undertaken exclusively from the P&O Portsmouth cruise-ferry, the Pride of Bilbao, which sails from Portsmouth (southern England) to Bilbao (northern Spain) traversing British, French and Spanish waters. The ferry follows a set route although the course alters from time to time. Effort-related sightings data were obtained from a total of 85 ICES (International Council for the Exploration of the Sea) rectangles (Each measures 15' latitude by 30' longitude (Fig. 1b).

Monthly seabird surveys were undertaken year-round from the Pride of Bilbao. Each return crossing extended over four days. In summer, this enabled the whole of the route to be sampled at least once during daylight hours and in winter approximately 75% of the route, the main gap in coverage then being the northern Celtic

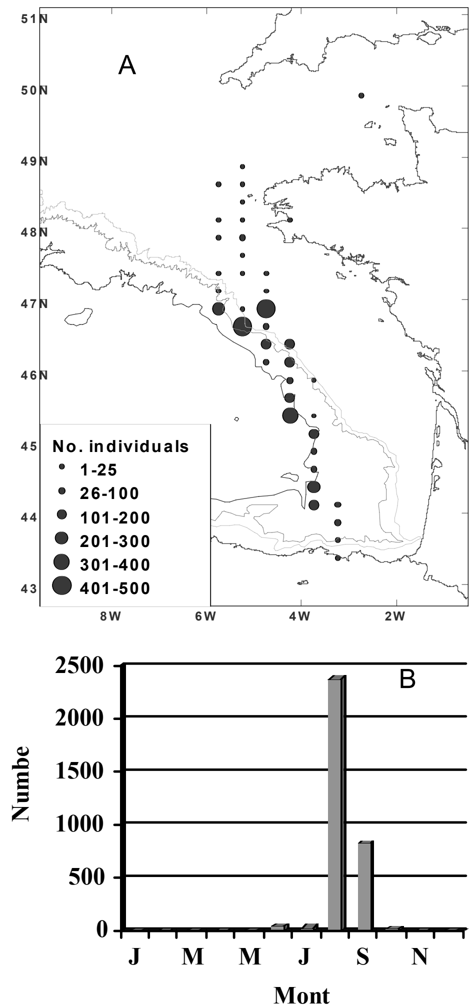


Fig. 2. Cory's Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000, (b) seasonal abundance.

Biscay shelf edge. Most survey trips were undertaken in the last week of each month.

On each trip, effort-related seabird recording was carried out during available daylight hours by three experienced volunteer observers. Recording was made from a fixed position on the bridge of the ship, at a height of 32m above the surface. The ship's speed was 15-22 knots. Each seabird was counted once only, sightings being grouped into minute-long periods. Between 1995 and July 2000, all birds were counted within an assumed strip-width of 4km, measured as 2km either side of dead ahead. For each seabird sighting the following recordings were made: species name, number seen, and where possible, age and sex. Notes on behaviour were also made including oiled birds and associations with fishing vessels and cetaceans.

From August 2000, refinements to the methodology were made to enable estimations to be made of bird density in the ICES rectangles. All birds ahead of the ship were recorded only once as being either inside an imaginary recording box, 300×300m, on the starboard side, or outside the box (but presumed within 2km). Effort data was collected simultaneously with sightings data, to enable the number of sightings to be scaled to recording effort and the calculation of relative frequency, abundance and density to detect change. At 15-30 minute intervals, or whenever the ships' course changed, a range of variables was measured, including the ship's speed and course, and sea and weather variables such as sea state and windspeed (Evans 1995).

Between 1995 and February 2001 63 (four-day) survey trips were made, resulting in more than 65 000km of completed search effort. Bird data were collected on 51 trips, totalling just under 50 000km of survey effort, coverage occurring evenly through the seasons (N=16-18 trips).

Using formulas in MS Excel spreadsheets, we calculated the number of km of survey effort travelled per ICES rectangle (30'latitude by 15' longitude) and the latitude and longitude of all bird sightings (from timings).

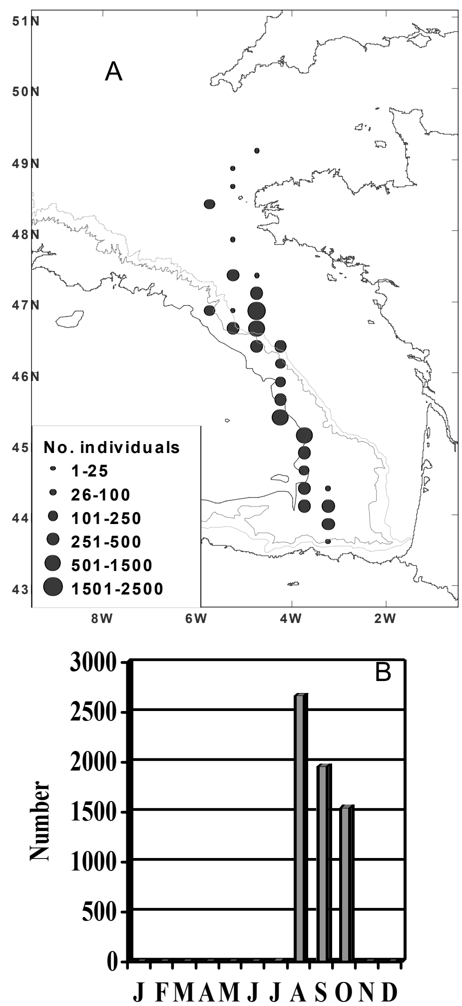


Fig. 3. Great Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000 , (b) seasonal abundance.

## 3. Results

### Cory's Shearwater.

Only four birds were recorded in March/April. Small numbers of birds were regularly seen in June/July, a total of 70 (2% of all birds) being recorded. Numbers increased markedly in August, 2386 (72%) being recorded. During September and October numbers 'dropped off', 831 (25%) and 21 (<1%) recorded respectively. Four (<1%) in November marked the seasonal limit of the species' inhabitancy of the bay (Fig. 2b). Cory's Shearwater was found mainly in areas of deep water >1000m deep and at areas of presumed upwelling to the north of the Celtic-Biscay shelf edge. Only seven (<1%) birds were recorded in the English Channel (Fig. 2a).

### Great Shearwater.

Great Shearwater was recorded from July-November. Small numbers were seen in July (10=<1% of total birds), followed by a major influx in August (2662=43%) and September (1955=32%) with numbers remaining high in October (1545=25%). Thereafter, a rapid drop off occurred, only two (<1%) being seen in November (Fig. 3b). The distribution of Great Shearwater was largely similar to Cory's that of Shearwater, the majority of records occurring in water of a depth >1000m, between the latitudes of 44°N to 47°N, a cluster of records occurring around the 4000m mark over the abyssal plain of the bay. Large numbers were also seen within 50km of the northern Celtic-Biscay shelf edge at presumed areas of upwelling, mainly in late October 2000. Only 4 (<1%) birds were recorded in the English Channel (Fig. 3a).

A comparison has been made between the two peak years (1999 and 2000) for large shearwater species (Fig. 4). This is of particular interest when considering the arrival and departure dates of each species. Cory's Shearwater arrived in large numbers in August of both years and then disappeared again by October. In 1999, Great Shearwater was present in large numbers in late August but had largely disappeared from the area by late

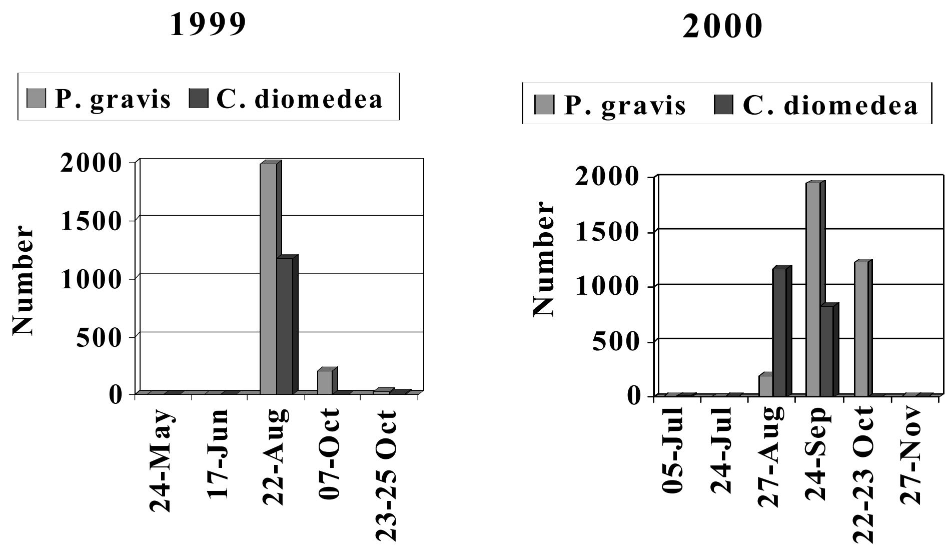


Fig. 4. Seasonal abundance of Great Shearwater and Cory's Shearwater in the Bay of Biscay in 1999 and 2000.

October, but in 2000, birds had only just begun to arrive in August and were then present in high numbers in both September and October. In the latter month high numbers were present unusually far north from 46°22'N to 46°57'N in waters of a depth <200m. On October trips, the ferry rarely reaches the northern end of the Celtic-Biscay shelf edge before dark and so numbers of Cory's Shearwater present may be much higher in August and September when daylight permits observations.

**Sooty Shearwater.**

Sooty Shearwater occurred in low numbers throughout the study area during the autumn months. There were records of single birds in February, April, July and December and the main peak occurred in August (18=15%), September (86=73%) and October (6=5%). There were small peaks evident just to the north of the northern Celtic-Biscay shelf edge around the canyons of the southern bay (Fig. 5a). However, in contrast to the two larger species, a substantial proportion of the total was recorded in the English Channel: 21 birds (18%) compared with 97 in the Bay of Biscay (Fig. 5b).

**Manx Shearwater.**

Manx Shearwater has a distinctly different seasonal distribution to the other shearwaters covered in this paper. Manx was seen mainly during the spring months March-June (*c*80% of birds) (Fig. 6b). During the autumn (July-October) only 38 birds (*c*24%) were seen. Furthermore, only 12 birds (*c*7%) were in water of depth >1000m, a high proportion of these being recorded in the shallow waters of the English Channel (*c*56%) (Fig. 6a).

**Mediterranean Shearwater.**

Records of Mediterranean Shearwater were fairly evenly spread through the survey area in small numbers (57 birds recorded), although only 8 (*c*14%) were seen in the English Channel). A small peak is noticeable around the Cap De Breton canyon in the south of the bay. Seasonally, there is a fairly wide spread of records, the peak period being June-November (*c*95% of birds). The concentration of records during October is probably not representative of the seasonal inhabitancy of this species and includes a single group of 16 birds (Fig. 7).

**Little Shearwater.**
It is difficult to come to any firm conclusions about the status and distribution of Little Shearwater in the survey area because of the small sample size. Only 10 birds were recorded, including 8 in August 1999, all of which were between 44°00'N [B4]and 46°05'N in or near to deep water areas. Eight birds were recorded over water of depth >4000m.

## 4. Discussion

This section will concentrate on comparing the previously-known status and distribution of the shearwater species under dis-

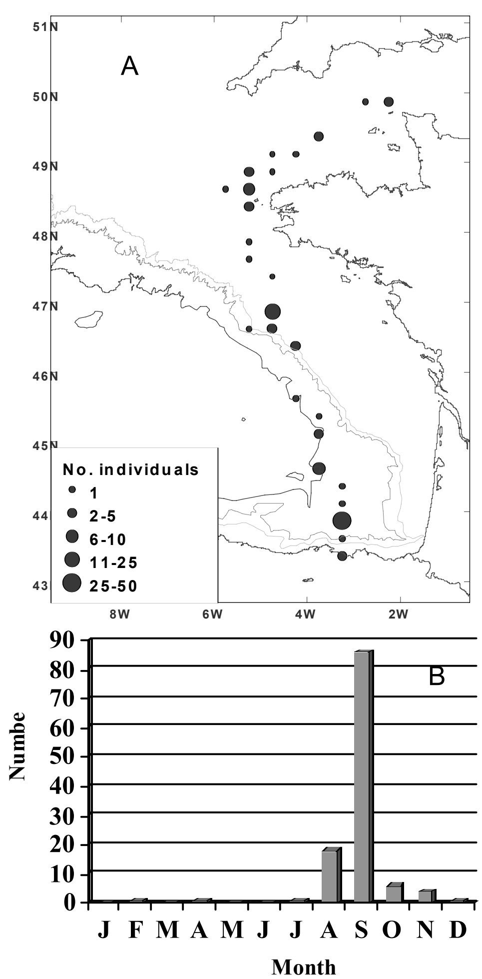


Fig. 5. Sooty Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000, (b) seasonal abundance.

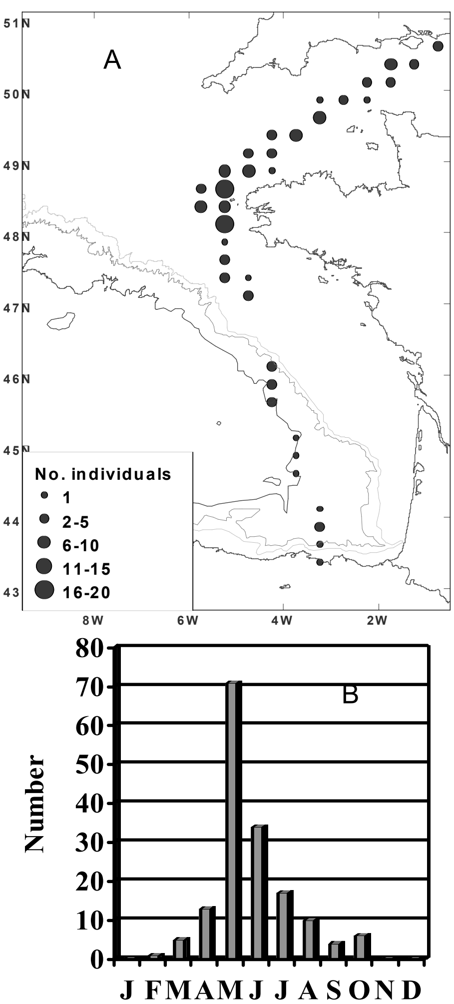


Fig. 6. Manx Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000, (b) seasonal abundance.

cussion in this paper with the results of the BDRP surveys of 1995-2001. At present we have insufficient data to comment on possible food sources for these species in the survey area although there is of course an intimate link between food and their presence.

Cory's Shearwater. Cory's Shearwater breeds only in the North-East Atlantic Ocean and the Mediterranean Sea, nominate *C.d. diomedea* nesting in the Mediterranean and the Atlantic race C.d. borealis on various islands between 15°N and 40°N. Birds are present in the breeding areas from late February until October-November when southbound trans-equatorial dispersal begins from all colonies to wintering grounds in the South Atlantic. Non-breeding birds are known to disperse across the Atlantic where they are found off the eastern coastline of North America in good numbers from July-October (Cramp 1977, Haney & McGillivary 1985). They also disperse into the Bay of Biscay, mainly into coastal waters, during the early autumn (Guerin 1999, Cramp 1977). Birds then seem to move into deeper water from August-October before southbound migration begins. Large numbers can be seen off Galicia (northwest Spain) from late August-November, a migratory peak occurring in October (Paterson 1997). Birds are rarely seen north of Irish waters (Bourne 1986, Skov *et al*. 1994, Stone *et al*. 1995) and are rarely found in waters of temperatures <13°C (Skov *et al*. 1994).

The results show that Cory's Shearwater is recorded infrequently in spring and early summer when it is present in large numbers in coastal Biscay (Guerin 1999) and then moves into deeper waters during August to late September before migrating south for the winter. We do not know whether the birds we recorded were *C.d. diomedea* or *C.d. borealis* because of the difficulty of separating these subspecies at sea. There are currently no records in the study area of 'Cape Verde Shearwater' *C.d. edwardsii* although it is possible that it may occur as its dispersal range is poorly known.

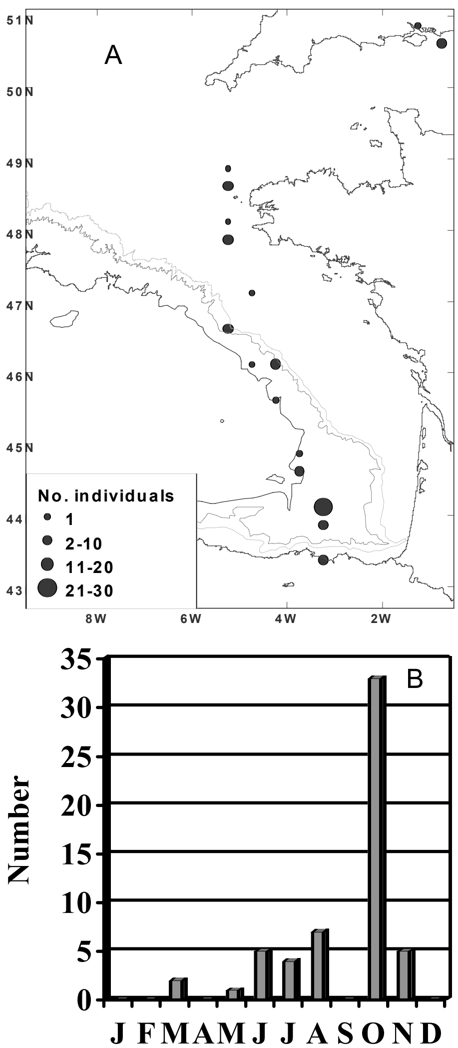


Fig. 7. Mediterranean Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000, (b) seasonal abundance.

Great Shearwater breeds only on four islands in the South Atlantic, principally on the Tristan da Cunha Group at an approximate latitude of 37°S, where *c*2 million pairs breed (Voous & Wattel 1963). The species is a trans-equatorial migrant, migrating north from April to June before congregating in offshore waters off eastern North America from June to August. Most breeding birds are back at their colonies by September (Rowan 1952). In the survey area, the peak numbers are seen from August to October and it is likely that most birds, if not all, are non- or pre-breeders that follow the prevailing west winds to feed in upwelling zones in the Bay of Biscay, and do not undergo a moult (Bourne 1986). It is also likely that most breeding birds return to the South Atlantic through a mid-ocean route (Bourne 1995), which may explain why numbers in the northeast Atlantic seem to be substantially lower than those in North American waters from June to August. Great Shearwaters appear to congregate only in eutrophic water zones, migrating rapidly through dystrophic regions (Voous & Wattel 1963).

Although the Bay of Biscay seems to be the most important site for this species in European waters, it is possible that it also occurs in high densities off the Irish shelf edge, the circumstantial evidence being the high numbers recorded sporadically off coastal watchpoints (Newell 1968). However, dedicated surveys in these waters have not produced high densities (Stone *et al*. 1995). Great Shearwater is certainly infrequent north of latitude 55°N except in the waters off southeast Greenland (Skov *et al*. 1994). It is probable that significant numbers also occur off the shelf edge around northwest Spain where large numbers are reported regularly during northwest winds off Galicia, up to 3000 birds being noted in a day (Paterson 1997).

In Biscay, numbers and arrival dates vary from year to year, but the reasons are unclear, although the availability of food certainly plays a large role in their occurrences on the other side of the Atlantic (Brown *et al*. 1981).

The only known breeding colonies of Sooty Shearwater in the South Atlantic are in the Cape Horn region and in the Falkland Islands (Philips 1963). As with the Great Shearwater, Sooty migrates north first into North American waters from May to -June, and at that time the two species are present in an approximate ratio of 100:1 in favour of Great Shearwater (Philips 1963). However, when they arrive in northeast Atlantic waters from late July to early October, the numbers of the two species are similar, counts in the thousands being recorded most years off the west of Ireland and very occasionally in the North Sea (Various observers *pers comm*). It has been suggested that all North Atlantic birds are non breeders or pre-breeders (Cooper *et al.* 1991), which would partly explain the disparity between records of Sooty and Great Shearwaters on each side of the Atlantic.

In the survey area, Sooty Shearwater is not particularly common (Fig. 5) and does not congregate in numbers in areas of great depth differential as the large shearwaters do. The species seems to prefer colder waters for feeding because the main concentrations in the northeast Atlantic are further north; *e.g.* Rockall Bank and the Faeroese fishing grounds (Cramp 1977, Stone *et al*. 1995).

Approximately 94% of the world's population of Manx Shearwater breeds in the waters of Britain and Ireland. The most important areas for this species are off the west coast of Scotland, the Celtic Sea and the waters off southwest Wales and around the Irish Sea coast (Stone *et al*. 1995).

In the survey area, most birds seen in spring were north of latitude 48°N in the southwestern approaches and English Channel. It seems likely that migrant Manx Shearwaters head straight to wintering grounds off the east coast of South America (Cramp 1977). Birds that have been observed in the deeper waters of the Bay of Biscay are likely either to be adults roaming widely (spring and summer) or young birds (autumn).

As their name suggests, Mediterranean Shearwaters breed only in the Mediterranean Sea. Their movements, as with most shearwater species, are imperfectly known. However there is evidence of a large moulting population of birds in coastal waters of the Bay of Biscay during post-breeding dispersal in June. The greatest numbers are found off Vendee in July and the Mor Braz area of Brittany in mid-August to early September, when small numbers are recorded regularly in the English Channel and North Sea (Le Mao & Yésou 1993). Most birds return to the Mediterranean Sea in September and October, although some winter off southwestern Spain and along the Atlantic coast of Morocco (Le Mao & Yésou 1993). The wintering range is inadequately known, partly because this species fairly recently was split from Manx Shearwater (Cramp 1977), and more recently still was split further into Yelkouan (yelkouan) and Balearic (mauretanicus) Shearwaters (Snow & Perrins 1998), although not all authorities agree. Because of the near-impossibility of separating the two latter species by sight, this paper continues to treat both as 'Mediterranean Shearwater'.

In the survey area the low numbers of individuals recorded makes it difficult to ascertain any definite trends as to seasonal distribution and status. Birds have been recorded fairly evenly throughout the March-November period, a peak occurring in late summer and autumn (especially October). Records are fairly well spread over the area, coming from all sectors. The best area for sightings seems to be over the Cap De Breton canyon in the southern Bay.

In the North Atlantic, probably fewer than 10 000 pairs of Little Shearwater breed on the Azores, Canary Islands, Madeira and Cape Verde Islands (Snow & Perrins 1998). They seem largely to be sedentary, which has encouraged their high degree of subspeciation worldwide (Cramp 1977). However they may be more dispersive than we realise. Records from British, Irish and even North American waters suggest that individuals may wander widely.

In the survey area, very small numbers were recorded during August-October but other observers have seen the species regularly from the ferry, sometimes up to 20 in a trip (Various observers *pers comm*). However, because of the difficulties of validating claims, we decided not to compare our data to that of these other observers.

*Acknowledgements*. We would like to thank the staff of P&O Portsmouth, particularly Penny Guy, and the Entertainments Crew of the Pride of Bilbao for supporting the project in many ways. Dr Peter Henderson and Dr Richard Seaby of Pisces Conservation helped generate the Excel spreadsheets used in analyses. Nigel Symes made valuable comments on the text

Tab. 1. Survey effort (km travelled) by season in the English Channel and Bay of Biscay on bird recording trips.

| Season (Months) | No of trips | English Channel km travelled | Bay of Biscay km travelled |
|---|---|---|---|
| Spring (Mar-Jun) | 17 | 8683 | 11912 |
| Summer/Autumn (July-Oct) | 18 | 5198 | 9830 |
| Winter (Nov-Feb) | 16 | 7338 | 5629 |
| Total | | 21219 | 27370 |

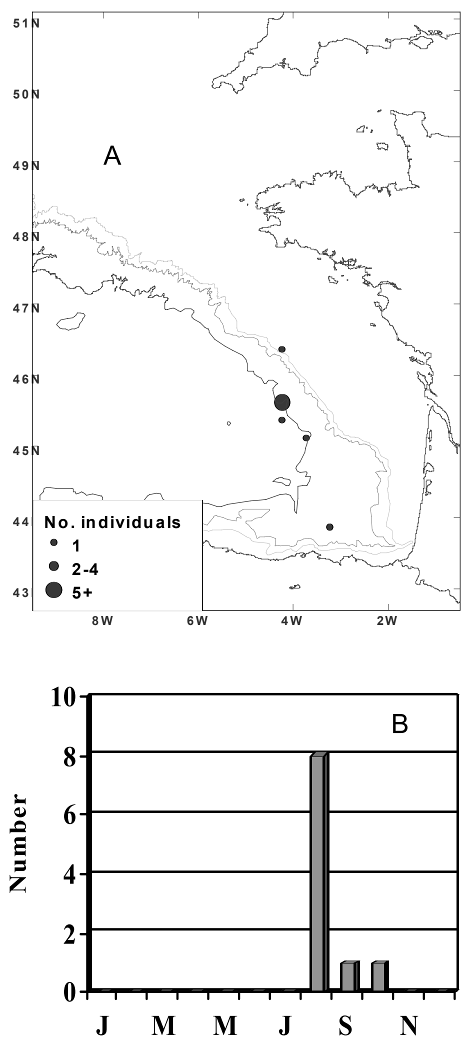


Fig. 8. Little Shearwater in the Bay of Biscay. (a) Distribution of 1995-2000, (b) seasonal abundance.

and supplied casual records to aid in the interpretation of BDRP data. For help with data entry, we would like to thank Nigel Bourn, Nigel Symes, Helen Williams, and especially Sally Taylor. Finally, we would like to thank the other BDRP surveyors for their recording efforts including Paula Bates, Dr Tim Melling, Russel Neave, Robin Plowman, Andy Schofield, Nigel Symes, Gordon Trunkfield and Rolf Williams.

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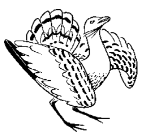
**Ornis Hungarica : 129-133. 2003**

# International corncrake monitoring

N. Schäffer and U. Mammen

Schäffer, N. and Mammen, U. 2003. International corncrake monitoring. – Ornis Hung. 12-13: 129-133.



An International Corncrake Monitoring Scheme is suggested in order to follow the population trend of Corncrakes affected by large-scale changes in land-use in Central and Eastern European Countries. A low cost and relatively low effort monitoring method is described in this paper. Corncrake experts in all breeding range countries are asked to support the project, co-ordinated by the International Corncrake Conservation Team (www.corncrake.net).

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## 1. Introduction

The Corncrake *Crex crex* breeds in Europe and eastwards through Central Asia as far as western China, and winters in sub-Saharan Africa. From recent surveys in Central and Eastern Europe and from new population estimates for Asiatic Russia, it can be shown that the Corncrake is considerably more numerous than was thought in the early 1990s. The global population is estimated as 1.7-3.0 million singing males, approximately 1.1-1.8 million of these being in Europe (Schäffer & Green 2001).

Historically, rapid declines have occurred in western Europe as a result of changes in agricultural practices. The main factor in the decline throughout western and central Europe has been the intensification of grassland management, which leads to earlier and more rapid mowing of hay and silage. The decline was continuous through the 20$^{th}$ century until the early 1990s, by which time Corncrake populations in western Europe were tiny. However, there are clear indications that in the 1990s the Corncrake population increased in several European countries. In central and eastern European countries, the apparent causes were political changes and privatisation, which had led to reductions in farming intensity and to land abandonment. Parallel increases in some western European populations are thought to represent an overspill from populations further east.

Corncrake specialists in Europe expect this increase to reverse rapidly and soon in the species' Eastern European strongholds, because, although land abandonment temporarily favours the species, abandoned areas rapidly become unsuitable through scrub encroachment. In central and eastern Europe, a likely alternative to land abandonment is the intensified management of hay meadows, or their conversion to arable use, changes which also would result in widespread habitat loss. For these reasons, the species is considered globally threatened and is classified as Vulnerable (BirdLife International 2000).

One of the main objectives of the International Corncrake Action Plan (Crockford *et al.* 1996) is monitoring the global Corncrake population. This paper presents the International Corncrake Monitoring Scheme (ICMS), which employs methods designed to minimise the effort required for its implementation, in terms of time and expenditure.

## 2. Methods

An annual count, always on the same sites, of Corncrake males forms the basis of the ICMS. Fieldworkers may select survey sites freely, provided that the survey plots are as typical as possible of the areas that Corncrakes occupy in that region or country. Field workers are also free to choose the size of their survey area, but it should be as large as possible. It is sensible that, in the main, they select sites that can be surveyed completely during a single night, but it is allowable to register larger sites, in cases when not only will several field workers be able to work together, but also it can be guaranteed that the large site can be surveyed over several years. In some countries, it might be preferable to appoint regional or national co-ordinators whose detailed knowledge of the region would allow them to select the sample sites and to supervise the survey work.

The survey results should be reported (on standardised survey forms) to us at the International Corncrake Conservation Team annually (see annex). In turn, we will report progress made and will record the results of the monitoring on our home page (www.corncrake.net) and in mailings.

### 2.1. Survey

#### *2.1.1. Timing and number of surveys*

In most countries, the period 20 May-10 July is suitable for counting singing males, although 1-30 June is best. Male Corncrakes are most likely to be calling continuously at night in June. The density of singing Corncrakes recorded in any single June nocturnal survey is probably *c*70-80% of the true average density of singing males present. Hence, a total of 2-3 visits is preferred, to reduce the chance of unreliable results from an anomalous survey carried out when males happen to sing less than usual. What is most important is that the number of surveys and their timing should be the same in every year.

#### *2.1.2. Survey method*

Male Corncrakes are most likely to be heard singing in the middle of the night from 2300 to 0200 local time and so we recommend that surveys be restricted to that time period. A small but variable proportion of Corncrakes sing during the day but usually not continuously. Hence, diurnal counts can give a very misleading underestimate of numbers, particularly when a series of such annual counts is considered: the figures obtained would bear little relationship to the real numbers of male Corncrakes. For obvious reasons, windy nights (wind stronger than force four) should be avoided. It is recommended that the site be visited the day before the survey, in order to plan the nocturnal survey route, which should always remain within 500m of any potential Corncrake habitat (*e.g.* meadows, pastures, nettle beds). During the nocturnal survey, fieldworkers should stop at the chosen points and listen. All Corncrakes heard should be recorded immediately on a map. The stopping places should be within 500m of every portion of suitable habitat. A fieldworker finding a Corncrake must record its direction from at least 2 separate locations and mark its position on a map by triangulation. Fieldworkers must never place too much reliance on the volume of the Corncrake call, because the volume depends on which way the bird is facing when it calls. It is all too easy to form the illusion of 2 birds (one nearby, the other distant) when a bird turns to call in another direction. Fieldworkers must also beware the sound reflections and echoes from rocks and buildings, for these give the illusion of 2 birds in different places. This problem can be resolved by the observer listening carefully to the calling rates; two real birds will call at different rates, but echoed calls from a single bird naturally are uttered at the same rate. It should be noted that the distance at which singing birds are detected can vary considerably between nights and between individual observers. Observers can adjust the separation distance between listening stops according to the prevailing conditions, wider separation being possible on still nights, but closer spacing will be necessary on breezy nights. Tape lures of Corncrake calls will bias the survey results, and therefore must not be used.

### 2.2. Interpretation of the results

During every survey, the position of the Corncrakes located should be marked in a map. At the end of the field season, data from each survey night should be summarised on a summary map that presents the location of every singing bird. If the records show that on separate visits singing birds were present at locations less than 200m apart, it is usually best to treat the two records as referring to the same bird, but records of birds separated by more than 200m should be treated as referring to two birds. However, if an area known to hold singing Corncrakes has been mowed, a neighbouring area may very well hold more singing birds than the previous visit some of may have moved more than 200m, and these will now be closer than 200m to other singing birds: judgement is therefore called for in assessing the numbers of individual birds. Although some degree of overestimate and underestimate is possible in these circumstances, the extent is likely to be small, and is most important it to apply the agreed rules consistently year to year.

### 2.3. Instructions for filling in the survey forms

The survey forms are largely self-explanatory, but those aspects that might lead to misunderstandings are elaborated below:

***Sheet A***

Sheet A needs to be completed only once for each site. A map showing the exact location and boundaries of the survey area should be attached to Sheet A. Essential points are:

- Name of survey area
  The name selected for the survey area should take into account either the local name or should mention the nearest sizeable human settlement. This name must be repeated on Sheet B.

- UTM quadrat or geographical co-ordinates
  Please fill in the number of the UTM quadrat or the geographical co-ordinates of the site. Should this not be possible, please attach a map, 1:500 000 scale or larger, on which the location of the survey plot and some of your country's borders are clearly shown.
- Number (No)
  Please ignore this line.
- Method
  Please tick to show whether or not you followed the recommended method, or describe your alterations. Once you have chosen your method, you must keep strictly to it in all survey years.
- Habitat
  Although you do not have to fill in this line, please make a rough note of the percentage of different vegetation types in the entire survey area during the calling period.

***Habitat types: some problems***

The ICMS selected a very wide range of habitat types, but despite this it is not always straightforward to identify habitat types with certainty, because many habitats do not fall clearly into any category mentioned on sheet A some of the main problems are:

- A spring hay meadow may be used as pasture later in the year
  In such a case please describe the habitat type existing during the main Corncrake breeding season.
- The term 'cultivated/uncultivated' is imprecise
  Please note that we have attempted to categorise habitats over a very large area (Europe, parts of Asia), and that uncultivated meadows and pastures (possessing features like rough topography, bushes, ditches, regular flooding) are distributed mainly in central and eastern Europe, having almost disappeared from western European countries.
- Corncrakes in clearfells
  Although this is a rather common observation in very large forest clearfells, particularly in eastern Europe, please remember that Corncrakes can use clearfells only for a short period, when tall herbs are dominant.

Please note that it is optional to complete the vegetation categories.

***Sheet B***

A copy of Sheet B has to be completed annually and has to be returned by 31 August. Observers are requested to fill in this form for their survey site whether they have found Corncrake in that year or not.

- Reference Number (No)
  Observers will receive their Reference Number from the international coordinators after the first survey year, and so they are required to complete this line only from the second survey year onwards.
- Survey results
  Observers are requested to fill in one line for each survey of a site during the season. For participation in the International Corncrake Monitoring Scheme, two to three surveys of a site during the year are sufficient. Should observers have carried out more than seven surveys of any single site, they should choose the results from no more than seven typical surveys.

- Best estimate of males present throughout the core season
  This is the most important information on the survey form. Observers are requested to fill in a figure of calling males present during the core breeding season based on their individual surveys and experience.
- Old data
  Should observers have old survey and monitoring data, they are requested to fill in an example of Sheet B for every year of these data. Old data are very valuable for our analysis of the population trend.

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**Ornis Hungarica 12-13: 135-141. 2003**

# Monitoring Corncrake *Crex crex* numbers in European Russia: the first stage

O. V. Sukhanova and A. L. Mischenko

Sukhanova, O. V. and Mischenko, A. L. 2003. Monitoring Corncrake *Crex crex* numbers in European Russia: the first stage. – Ornis Hung. 12-13: 135-141.

The Corncrake is a widespread and common species in Russia. Because agricultural conditions in Russia are unstable, it is important to establish the Corncrake's current population dynamics through studies in pilot areas. Two types of repeated censuses have been carried out:
1. In 1994, there were censuses on pilot plots in 4 regions. These regions had been censused between 6 and 18 years previously, but researchers had used a variety of methods.
2. From 1995 to 2000, regular nocturnal censuses were carried out, using standard methods in pilot areas in 5 'key territories' holding Corncrake concentrations.
The census years experienced varying weather conditions, such as spring flood levels and differing hay mowing sequences. The data obtained show that in the 1980s and 1990s in European Russia there was no tendency for Corncrake numbers to reduce, something that had been noted from the 1950s to the 1970s. It is extremely important that the present monitoring scheme continues and that a network be established of model areas in Russia to monitor Corncrake numbers annually and to track the changes in agriculture technology.

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## 1. Introduction

In the long history of ornithological science in Russia, very little attention has been paid to studying Corncrake. Its breeding range is large in Russia, and it was assumed to be common everywhere except in peripheral zones and to have a favourable conservation status; consequently it was not regarded as deserving special monitoring of numbers or dynamics. Specific surveys of Corncrake numbers in European Russia in 1995-1996 (Mischenko & Sukhanova 1999) have allowed the determination of the present and recent population sizes. However, to assess trends accurately, it is necessary to determine Corncrake numbers in the past and to establish a network of monitoring areas. For the future development of complex pro-active conservation measures for Corncrake, it is very important to know the history of Corncrake numbers and dynamics, initially in 'key areas' holding concentrations of breeding birds. The main aim of this paper is to assemble the first results of monitoring Corncrake numbers in several areas of European Russia.

## 2. Study area and methods

In 1994, censuses of calling male Corncrakes were carried out during the breeding season in pilot areas in 4 regions of European Russia. In all these areas, counts had been made previously in the 1970s or 1980s. In the Vologda Region, censuses were carried out in the Kharovskiy District (60°15`N, 40°10'E). In 1970-76, part of the surveyed area had consisted of small, partly boggy meadows overgrown with bushes either in a mosaic of fields or between 'islets' of forest and bushes. Another part of the area consisted of meliorated meadows. From 1975 into the 1980s, the total area of meadows increased significantly after further melioration, which grubbed out forest and bush 'islets', producing a marked reduction in field areas.

In the Kostroma Region, the study area is in the Manturovskiy District (58°10'N, 41°20'E) and is bounded by the Unzha River in the southeast and by woodland in the northwest. 81% of the square has elevations 40-50 m above the river level, 11% of which are flood meadows. In the Novgorod Region, the census was carried out near the township of Lyubytino (58°50'N, 35°25'E) on dry grass/herb meadows, most of which are on elevated ground. In the Moscow Region, the census was carried out in the Lotoshinskiy District (56°15'N, 35°50'E) on `meadows in the valleys of the rivers Bolshaya Sestra and Lama. Three areas possessed mainly flood-plain meadows (Solotcha, Dedinovo and Klyaz'ma), one (Zavidovo) had only dry meadows, but another (Ilmen) held both types. Solotcha was censused several times per season, but for comparisons only the maximum numbers are taken. In the other 4 pilot areas the censuses were conducted only once per season, in the same way every year. The analysis of the counts data was carried out with the TRIM 3 software (Trends and Indices for Monitoring data). The squares of all surveyed areas are represented in the tables below.

One of the main aims of this research was to determine Corncrake population trends in the pilot areas, and in order to make valid comparisons with earlier data, we could not use unified methods, but had to use the earlier methods. In all pilot areas, the census of calling males was carried out during the breeding season. Using data from repeated censuses, below we present the highest density figures obtained.

In the Vologda Region we used 2 survey methods: route and plots. We employed both once-only and repeated surveys, but without keeping to the exact locality. During a route census, the birds were recorded in a strip equal to the mean distance of discovery (average 150 m+150 m=300 m). For censuses on plots, the points where calling males had been recorded were mapped. Surveys took place in early morning (0300-0900) or in late evening (2100-2300). The 1970-1976 and 1994 surveys were carried out from 10-30 June. In 1994, the following technique was employed on large meadow squares; three people, moving parallel to each other at 50-70 m intervals, would count Corncrakes at the same time. The total width of the survey strip was 300 m.

In the pilot area in the Kostroma Region in 1982-1985, Grabovsky (1993) mapped calling males on large plots, where constant routes were followed during the night and in daylight (during the seasonal peak of vocalization). From 13-15 July 1994, the census used Grabovsky's mapping technique and counting methods, being repeated at night (2300-0400). In the Novgorod Region in late May and early June 1984, E. S. Ravkin carried out a bird route census that included the Corncrake. Each route in the census was 6 km long. The surveys were made in early morning and late evening according to Y. S. Ravkin's methods (Ravkin 1967); the results were extrapolated to produce numbers per km$^2$, based on the mean discovery distances, by the formula:

$$K = \frac{40a + 10b + 3c}{Nkm},$$

where

*K* - the number of individuals per km$^2$

*a* - the number of individuals discovered at a short distance from the observer (up to 25 m)

*b* - the number of individuals in the middle distance (25-100 m)

*c* - the number of individuals at a far distance (100-300 m).

In 1994, from 5-8 June, we repeated the surveys using the same method.

In 1987, 1988 and 1994 plot censuses were carried out in the Lotoshinskiy District of the Moscow Region. Furthermore in 1994, a route survey was made on a strip 500 m+500 m wide and 1.9 km long. These censuses were carried out at least 3 times per season (from late May to early July) in the early morning (0400-0800) and late evening (2100-2359) hours.

From 1995-2000, for the first time in European Russia, we conducted repeated night censuses using standard methods with pilot areas in 5 'key territories' (*de facto* IBAs) holding Corncrake concentrations:

1. Solotcha flood plain, the left-bank area of the Oka River valley (Ryazan District of Ryazan Region, 54°48'N, 39°47'E).
2. Klyaz'ma flood plain (Vladimir Region, 55°58'N, 39°30'E).
3. Dedinovo flood plain, the left-bank area of the Oka River valley (Lukhovitsy District of Moscow Region, 55°10'N, 39°18'E).
4. Zavidovo Reserve (Konakovo District of Tver Region and Klin District of Moscow Region, 56°27N', 36°17'E).
5. Ilmen Lake Lowland. Surveyed area was located on the northwest bank of the Ilmen Lake (Novgorod District of the Novgorod Region, 58°23'N, 31°02'E), 20 km southwest of the city Novgorod.

## 3. Results

### 3.1 Analysis of literature

The first step in the determination of Corncrake past and present distribution and abundance in European Russia was the analysis of literature on this species. However, the specific data on Corncrake numbers or population density are very fragmentary and are presented in but a scattering of papers published after 1945. It is difficult to make comparisons, because the authors had used different survey methods that were poorly described.

The sources from the late 19$^{th}$ century to the early 20$^{th}$ century show that Corncrake was a common or abundant species all over European Russia (except at its northern range boundaries), being most abundant on flood meadows. Without producing any quantitative data, the authors often gave very picturesque descriptions of this species' great numbers. Therefore, Bogdanov (1871) noted, 'the flood meadows of the Kama and Volga

Tab. 1. The results of the repeated Corncrake censuses with the Intel apse in several years in four regions of European Russia.

| Regions | 1970-1976 | | 1982 | | 1983 | | 1984 | | 1985 | | 1987 | | 1988 | | 1994 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens. | km$^2$ | dens |
| **Vologda** | 3.4 | 3.8 | | | | | | | | | | | | | 5.8 | 3.8 |
| **Kostroma (flood-plain)** | | | | | | | | | 36.0 | 0.8 | | | | | 36.0 | 1.5 |
| **Kostroma (dry meadows)** | | | 5.4 | 3.1 | 5.4 | 2.6 | | | 5.4 | 1.5 | | | | | 5.4 | 2.4 |
| **Novgorod (dry meadows)** | | | | | | | 1.1 | 7 | | | | | | | 0.9 | 1.1 |
| **Moscow** | | | | | | | | | | | 3.7 | 2.4 | 3.3 | 3.6 | 3.9 | 3.3 |

Rivers are the real Corncrake kingdom. It is also numerous in all large and small river valleys of the Volga basin. It lives in the Ilovlya, Medveditsa, and Khoper River valleys in great numbers'. Zhitkov & Buturlin (1906) wrote that the air on the meadows of the Simbirsk (now Ul'yanovsk) Region was 'literally filled' with Corncrake calls.

In the Moscow Region, the most populated and developed in Central Russia, a marked decrease of numbers was noted in the 1930s (Ptushenko & Inozemtsev 1968). Spangenberg & Oliger (1949) pointed out a decrease in Corncrake numbers in the Darwin Nature Reserve at Rybinsk Reservoir, already considering it as a rare species. According to these authors, in 1946 Corncrake numbers on the meadows of the Reserve had reduced to only 1 male per 2-3 km, whereas before the reservoir had been built this species had been abundant there (Isakov 1949). At the same time, Nemtsev (1953) notes Corncrake as a common species of the Darwin Nature Reserve, noting that it inhabits not only meadows but also tall weed growth at former village sites. In the other regions of European Russia such as Perm (Vorontsov 1949), Nizniy Novgorod (Vorontsov 1967) and Leningrad (Malchevskiy & Pukinskiy 1983) no marked decrease of the numbers was observed until the early 1960s.

From the 1950s in European Russia, mechanized mowing became widespread, the first dates of mowing becoming earlier. At the same time, extensive ploughing up and draining of meadows took place. These events had an adverse effect on the Corncrake population. In the Moscow Region, an abrupt decrease in numbers began from 1954-1955 (Ptushenko & Inozemtsev 1968). A severe decrease in numbers in the Mary-El Republic became noticeable from the early 1960s (Baldaev 1973), and in the Leningrad Region and Mordovia from the late 1960s or early 1970s (Malchevskiy &

Tab. 2. Dynamics of Corncrake numbers (calling males) in the pilot areas in 1995-2000.

| Names | S | 1995 | | 1996 | | 1997 | | 1998 | | 1999 | | 2000 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | km$^2$ | n | density | n | density | n | density | n | density | n | density | n | density |
| **Solotcha** | 3.4 | - | - | - | - | 42 | 12.4 | 50 | 14.7 | 91 | 26.8 | 81 | 23.8 |
| **Klyaz'ma** | 4.7 | 64 | 13.6 | 29 | 6.2 | 33 | 7.0 | 66 | 14.0 | 77 | 16.4 | 79 | 16.8 |
| **Ilmen Lake** | 6.6 | 21 | 3.2 | - | - | - | - | 67 | 10.2 | - | - | | |
| **Zavidovo** | 21.1 | 99 | 4.7 | - | - | 108 | 5.1 | - | - | - | - | | |
| **Dedinovo** | 16.7 | 156 | 9.3 | 233 | 13.9 | 140 | 8.4 | - | - | - | - | | |

Pukinskiy 1983, Lugovoy 1975). A two-fold decrease in numbers was noted on the flood plain of the Klyazma River during the period 1970-1982 (Izmailov & Salnikov 1986).

### 3.2. The current agricultural crisis and its possible influence on Corncrake

Towards the end of the 1980s, a deep and prolonged agricultural crisis began in Russia, and still continues. Because of deficits in fuel and spare parts, mowing now begins later in the year. Ministry of Agriculture statistics show that, in forest zone regions, previously before 6 July each year up to 23% of areas of sowed and natural herbs had been mowed (now, it is usually only 15-20%). Even in the drought year of 1999 in the most developed Moscow Region, this parameter has not exceeded 38%. At the end of the 1980s timing of mowing in the forest zone had changed to peak in the first half of July, which obviously had increased the mortality rate of Corncrake chicks. In Russia, pastures with low livestock densities can hold successful Corncrake populations (Mischenko & Sukhanova 2000). As livestock numbers began to decreased strongly (by 50% from the 1980s to 1998 [Agriculture in Russia 1998]), their pastures became much more suitable for Corncrakes.

As a whole, Russian production of pesticides (including herbicides and insecticides) reduced by a factor of 7.4 between 1986 and 1995 (Agriculture in Russia 1998). No generalized data are available for more recent years, but it is known that pesticide production (and accordingly, its use on fields) has decreased even more. Consequently, cereal and fodder crop fields recently have become important post-breeding habitats for Corncrakes. Visual observations and radio-tracking results have produced confirmatory data. If we include the total optimum area of fields right up to the limits of the Corncrake range in Russia, it is possible to estimate the actual increase in the species' post-breeding habitat area.

### 3.3. Results of the repeated censuses

Results of censuses carried out at an interval of between 6 to 18 years are represented in Tab. 1. Data from the surveys in the Vologda Region from 1970-1976 are presented as the mean of the long-term results. Pilot area censuses in 4 regions have revealed that only the Novgorod Region suffered a significant decrease in numbers. At present, we cannot explain this decrease in numbers; certainly there were no landscape changes. Corncrake numbers in the other pilot areas were stable, or had increased a little, but the increases were within the limits of annual fluctuations.

The results of repeated night censuses between 1995-2000 are represented in Tab. 2. These censuses included years that experienced different weather conditions, spring flood levels and hay mowing periods. However, we did not find an instant correlation between Corncrake density one year and the next and one or more of these factors. Possibly local soil humidity is very important. For example, both 1997 and 1999 were very dry, but in 1999, the spring flood level was appreciably higher. The continual censuses during 1995-2000 allow the possibility of a slight increase in numbers in the Solotcha and Klyaz'ma floodplains, allowing for annual number fluctuations. In spite of the reduction in Lyubytino, in the Ilmen Lake Lowland (also located in the Novgorod Region), a significant increase in numbers was recorded.

## 4. Discussion

The completed survey is the first step in establishing Corncrake numbers in Russia. Certainly, it would not be valid to reach categorical conclusions on long-term Corncrake dynamics based on such limited data. However, it is evident that between the 1980s and 1990s in European Russia, there was no decreasing trend in Corncrake numbers as noted in between the 1950s and 1970s. Probably we have seen increases in Corncrake numbers and density over the whole of the centre of European Russia.

The TRIM 3 analysis of the count data from Tab. 2 shows a significant increase in numbers in 1995-2000. The overall slope is 1.21, which implies a 20%, increase per year during this period. However, we have insufficient data to confirm this.

Due to the prolonged agricultural crisis, we have observed two different processes: the abatement of agricultural pressure on Corncrake habitats (positive influence), and agricultural land abandonment, which results in encroachment of habitats by bushes and forests (negative influence). Without special monitoring, we cannot predict the trend changes in Corncrake numbers. It is therefore extremely important to continue the monitoring process that has been established in Russia, by maintaining the network of model areas for monitoring Corncrake numbers annually and by tracking changes in agricultural technologies. It is essential that the selection of monitoring areas throughout the Russian regions should be both representative and differentiated. The goal is to identify those areas typical of Corncrake habitats that qualify as major Important Bird Areas (IBAs). No significant changes have occurred on surveyed meadows in the Novgorod, Kostroma and Moscow regions since the censuses began.

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**Ornis Hungarica 12-13: 143-149. 2003**

# Status of the Corncrake *Crex crex* as an indicator of biodiversity in eastern Hungary

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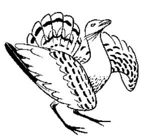
Wettstein, W. and Szép, T. 2003. Status of the Corncrake *Crex crex* as an indicator of biodiversity in eastern Hungary. – Ornis Hung. 12-13: 143-149.



In eastern Hungary during the summer of 1997, 17 natural grasslands holding Corncrakes were analysed to establish their general biodiversity and to compare them with neighbouring grasslands that lacked calling Corncrakes. The diversity of plants and birds on a small scale was measured through a series of plot and point counts, while the diversity at site level was measured by standardised counts of species richness per site for plants, butterflies and birds. Sites with Corncrakes did hold more plant, butterfly and bird species than the control sites. The small scale of the point and plot counts meant that we could not explain this difference, which may indicate that the effect was mainly caused by a more heterogeneous habitat structure at the Corncrake sites. This is supported by the fact that the number of indicator species of hedgerows, bushes and isolated trees, but not of indicator species of open grasslands, was higher at the sites with Corncrakes. Tree Pipit *Anthus trivialis*, Common Cuckoo *Cuculus canorus* were associated with the Corncrake, (Corn Bunting *Emberiza calandra* tended to be associated with Corncrake) while (Eurasian) Skylark *Alauda arvensis* and Yellow Wagtail *Motacilla flava* tended to be commoner at the control sites. Corncrakes can be considered as a good umbrella species for the biodiversity of wet grassland habitats because the large and heterogeneous habitats suitable for this species also provide suitable niches for most other species of wet grasslands breeding in eastern Hungary.

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## 1. Introduction

The Corncrake *Crex crex* is a globally threatened bird species living mainly in extensive tall grasslands. It is classified as 'Vulnerable' at both world and European level due to the long-term and very steep population decline of the species across its range. Breeding occurs in 34 countries but has been declining in Europe since the last century, on average since about 1990 by some 20% to 50% (Crockford *et al*. 1996, Tucker & Heath 1994). However, very recent population recoveries throughout Europe demonstrate also the potential for fast population increase under good conditions (Stowe & Green 1997).

In many countries the establishment of conservation programs for the Corncrake is still a difficult task because of its strong dependence on tall grasslands, which are also important in agricultural production (Green *et al.* 1997). This result in a distribution in Europe negatively correlated to the intensity of agricultural production (Green & Rayment 1996). To convince landowners as well as politicians to carry out Corncrake conservation programs, economically acceptable management

Tab. 1. Differences in species numbers of plants, birds and butterflies at plot level (a-diversity) and site level (b-diversity) between grasslands with Corncrakes (+sites) and control sites (-sites). sd = standard deviation. Significant differences are marked in bold letters. One-way ANOVAs.

| | Plant species | | | Butterfly species | | | Bird species | | |
|---|---|---|---|---|---|---|---|---|---|
| | +sites | -sites | F | +sites | -sites | F | +sites | -sites | F |
| α-diversity | 6.1 | 6.4 | 0.1 | -- | | | **6.2** | **5.5** | 3.4 (*) |
| sd | 2.6 | 1.9 | | | | | 0.9 | 1.0 | |
| β-diversity | **65.9** | **54.9** | 9.6 ** | **11.3** | **7.1** | 9.9 ** | **17.1** | **13.9** | 4.5 * |
| sd | 9.3 | 10.9 | | 4.7 | 2.6 | 3.8 | 4.1 | | |

(*) p>0.1, * p < 0.05, ** p < 0.01

plans (Schäffer & Weisser 1996) and further political arguments about the importance of Corncrake protection in nature conservation have to be developed.

It has therefore been suggested, as an important aspect in Corncrake conservation research, to analyse the status of the Corncrake as an indicator of biodiversity (Crockford *et al.* 1996). It has long been known that Corncrakes inhabit areas of generally high biodiversity (Flade 1991), but no quantitative data are published that compare the diversity, in the same geographical region, between sites holding Corncrakes and sites where the birds do not occur in grasslands. We therefore analysed biodiversity aspects in a metapopulation of approximately 200 Corncrakes in eastern Hungary. The birds were dispersed in small groups over many isolated sites of natural grasslands situated in the plain of the Upper Tisza, and it was possible to compare diversity between occupied and unoccupied sites. Specifically, we wanted to analyse whether grassland sites occupied by Corncrakes also held a higher diversity of plants, butterflies and birds, if there were differences between measures of diversity at different scales, and if some bird

Tab. 2. Comparison of bird species numbers between sites with Corncrakes (+sites) and control sites (-sites), with species separated into five ecological groups. Significant differences are marked in bold letters. One-way ANOVAs.

| Variable | Nr. of species (means) | $R^2$ | n | F |
|---|---|---|---|---|
| Indicators of grassland | +sites: 4.5<br>–sites: 4.5 | 0.001 | 34 | 0.02 |
| Indicators of bushes, hedgerows and old isolated trees | +sites: **4.8**<br>–sites: **2.9** | 0.15 | 34 | 5.58 * |
| Other possibly breeding species | +sites: 2.4<br>–sites: 2.0 | 0.01 | 34 | 0.39 |
| Raptors | +sites: **2.1**<br>–sites: **1.4** | 0.10 | 34 | 3.49 (*) |
| Visitors | +sites: 3.5<br>–sites: 3.2 | 0.01 | 34 | 0.33 |

** = p<0.01, (*) = p<0.1

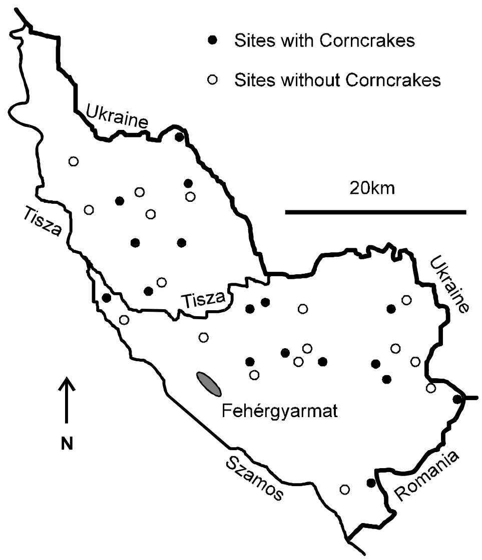



Fig 1. The location of the 34 study sites in the Szatmár-Bereg region of eastern Hungary (48°00'N, 22°30E).

species or groups of species were associated with Corncrakes.

## 2. Methods

The study area was the Szatmár-Bereg lowland (Important Bird Area HU35, category A1, Nagy 1998) in easternmost Hungary in a triangle bounded by the rivers Tisza and Szamos and the border with the Ukraine and Romania (48°00'N, 22°40'E), an area of approximately 1170 km$^2$. A general survey of the Corncrake was carried out in this region between 15 May and 12 June 1997, covering 137 km$^2$ of natural grassland.

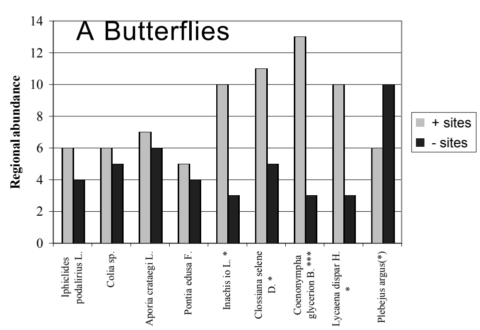
Based on this survey, we selected 17 clearly delineated grassland sites each holding at least two singing males so that a diversity analysis could be carried out. Additionally, for each site holding Corncrakes, we wanted to analyse (as a control) a clearly delineated natural grassland lacking calling males. To ensure a similar geographical distribution of the two site groups, for each Corncrake site we selected the closest properly surveyed site that lacked calling males. Because of the number of unoccupied sites in the north was limited, the separations between the pairs of site types are greater there. In the end, we had a total of 34 grassland sites in which we could compare the diversity of those that held Corncrakes with those that were unoccupied (Fig. 1). A second survey was made between 26 June 1997 and 6 July 1997 to establish presence of the birds. An important outcome of the second survey was that it confirmed the status of the study sites for the diversity analysis. At all occupied sites, Corncrakes were heard again, while none could be found at any of the control sites.

We used the MapInfo v4.0 GIS software for measuring the size of the potential breeding habitats, by identifying the border of the habitats on the digitised and geocoded 1:25 000 map and calculated the area of the identified polygons. There was no difference found in the extent of the area of occupied and control sites but the grasslands differed mainly thus: occupied sites had taller but less dense vegetation and comprised a higher degree of additional habitat structures, such as like bushes and old riverbeds (Wettstein, *et al.* 2001).

To measure the diversity of plants in the grasslands on a small scale ($\alpha$-diversity), we determined five points along a 500 m transect. The transects at Corncrake sites were placed through the centre of the area of highest density of calling males, because many of the occupied sites also contained large areas lacking Corncrakes . We had to be sure to sample habitats that

Corncrakes had preferred. At the control sites, transects were placed randomly. At the determined points, we placed a 33×33 cm frame in the vegetation and counted the number of plant species growing within the frame.

To measure diversity of birds on a small scale, a one-minute point count was carried out at every point selected for the plant analysis. Only birds within the borders of the grassland site were counted. For plants and birds, we calculated the average of these five counts respectively



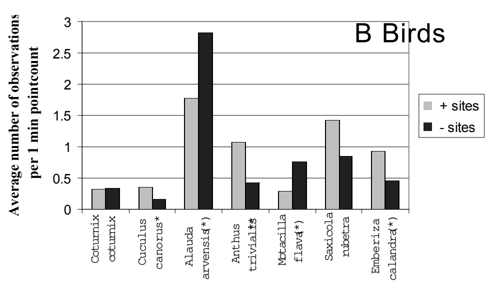




Fig 2. Comparison between sites with Corncrakes (+sites) and sites without Corncrakes (-sites) for: a. The regional abundance of the butterfly species that have been observed at 10 to 20 sites during all visits (Chi-square tests); b. The average number of observed bird individuals during the five one-minute point counts for all the observed indicator species of grasslands and for Cuckoo (One-way ANOVAs).
*** = $p<0.001$, ** = $p<0.01$, * = $p<0.05$, (*) = $p<0.1$

to compare the small-scale diversity between Corncrake sites and control sites. The density of butterflies was very low at most sites, so that no measure of diversity was established from the point counts. At the selected plots, mostly the very common species *Pieris sp.*, *Maniola jurtina* and *Coenonympha pamphilus* were observed, other species were only very rarely seen.

To obtain a measure of plant diversity at site level (β-diversity), the number of plant species that could be identified during a half an hour walk at constant speed was noted. The walk followed a course chosen to cover all the important grassland types present at the site proportionally to their area. The diversity of birds and butterflies at site level was measured as the number of species that could be found within the grassland during the site visit (approximately 3 hours per site). On any one day, one site of a pair would be investigated after 0800 and one in the afternoon before 1700 with occupied and control sites distributed equally at both times. The bird species list for each site was then divided into the functional groups 'Indicators of grassland', 'Indicators of bushes, hedgerows and old isolated trees', 'Raptors', 'Other species possibly breeding at the site' and 'Visitors' depending upon the breeding biology of the observed species.

## 3. Results

### 3.1. Plants

Grasslands holding Corncrakes had very variable plant diversity at plot level. Corncrakes occupied not only grassland with the lowest species diversity dominated mainly by *Alupecurus pratensis* (averaging only 2.6 plant species per 33×33 cm square) but also the most diverse alluvial meadow (11.4 plant species per 33×33 cm square). Species numbers at the control site plots were less variable, averaging 6.4 species per plot, very similar to the Corncrake site values. In the half-hour transect counts, however, which represent the large-scale diversity of the sites, the Corncrake sites held on average ten more species than the control sites (Tab. 1).

**3.2. Butterflies**

Species richness of butterflies at site level ranged from 3 to 19 species in Corncrake sites, averaging 11.3 species per site. In the control sites however, fewer species (ranging from 3 to 15) were found, an average of 7.1 species per site (Tab. 1). *Inachis io, Clossiana selene, Coenonympha glycerion* and *Lycaena dispar* occurred significantly more frequently at the occupied sites than at the control sites (Fig. 2a).

**3.3. Birds**

The bird point counts at the Corncrake sites produced an average 6.2 species per one-minute point count. In the control sites, the average tended to be lower at with 5.5 species per site. For each site as a whole, Corncrake sites had 17.1 species, averaging 20.5% more species than the control sites (Tab. 1). Analysis of the ecology of the observed bird species revealed that the higher diversity attained on the occupied sites was due mainly to the 'indicators of bushes, hedgerows and isolated trees' (14 species in total). On average, occupied sites held two more indicator species than did the control sites (Tab. 2). Furthermore, Corncrake sites were likely to hold one more raptor species than control sites (10 raptor species in total). However, we could not detect any difference between site types in the number of grassland indicator species (a total of 6 species). In addition, other species as possible breeders or visitors were found in similar numbers at both site types.

The comparison of the abundance of single species at the point counts shows differing results for different species (Fig. 2b). Quail *Coturnix coturnix* was observed in similar numbers at the two site types. Typical short-grass habitat species like Skylark or Yellow Wagtail were commoner in the control sites. The Cuckoo, the Tree Pipit and the Corn Bunting, however, were more often observed at sites with Corncrakes. We had classified both the Tree Pipit and the Corn Bunting as grassland birds, but both like bushes and isolated trees in their habitat, which is consistent with the result from the previous paragraph. Interesting patterns were further observed for the River Warbler *Locustella fluviatilis* and the Grasshopper Warbler *L. naevia*, which were both observed only at Corncrake site point counts (four occasions each). All other species did not show clear numerical differences between the two types of grassland or were observed on too few occasions to draw meaningful conclusions.

## 4. Discussion

The use of simple and fast field-methods makes a comparison to other studies difficult. Therefore, we cannot compare the biodiversity found in the Corncrake habitats of eastern Hungary to other similar habitats elsewhere. Our results reflect the features of a habitat choice within a region. Because the Corncrake preferred very diversely structured grasslands, it was also associated with a high general biodiversity. In the study region such diversity includes, in particular, thorn bushes on pastures, willow bushes, former riverbeds, hedgerows along ditches or paths and isolated old trees, which are highly important for other wildlife. These features mostly derive from extensive land-use over a long period, but in at least four sites, the observed diverse landscape structure is the direct consequence of cessation of land-use since the late 1980s. The fact that no difference was found between each site type for the number of generalist species probably breeding in them indicates that both kinds of grassland have similar basic ecological conditions, but differ in special features that affect the specialist species, such as site management or vegetation structure.

The region has large unprotected valuable areas, a circumstance that demands timely conservation action in devising and applying suitable methods of selecting key additions to enlarge the protected areas. Without doubt, Corncrake distribution is a good indicator of valuable wet grassland habitats. Its need for large heterogeneous grasslands and the indication of high β-diversity make it a good umbrella species for the grasslands of fluvial plains.

So far, no management conflicts with other threatened species have arisen from Corncrake-friendly management. The two species that were found to be commoner in sites without Corncrakes are both very common and prefer short-grass steppe-like habitats. This is a reminder that for dry habitats, completely different aspects have to be considered. It is also important to remember that such things as small patches of rich or diverse habitat untypical of the surrounding area may easily overlooked in this single-species approach, yet these patches may be very valuable for plant or insect populations (Simberloff 1998).

Further studies are needed to determine direct interrelationships between the occurrence of Corncrakes and additional landscape structures in grasslands (*e.g.* recognition of suitable habitat types during return migration and shelter or rest areas). However, if the Corncrake is to be regarded as an umbrella species for general conservation of biodiversity, these elements need special consideration. If the conservation approach also embraces the maintenance of the heterogeneous landscape structure associated with Corncrake areas, it will support a wide range of biodiversity at the same time. Isolated single-species management focused solely increasing Corncrake populations would not justify its status as an indicator species of biodiversity, because in very poor species habitats, management efforts may also boost population numbers (Simberloff 1987). To save the Corncrake and the high general biodiversity of wet grasslands, agricultural policy should encourage a variety of low intensity land-use practices, thus safeguarding the traditional flood plain landscape structure; with special soil relief and other structuring elements.

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**Ornis Hungarica 12-13: 151-156. 2003**

# Recent increases in numbers and the future of Corncrake *Crex crex* in Latvia

O. Keišs

Keišs, O. 2003. Recent increases in numbers and the future of Corncrake *Crex crex* in Latvia. – Ornis Hung. 12-13: 151-156.

Night counts of calling Corncrakes were carried out on 34 survey routes in Latvia during the breeding seasons of 1989-2000. The average breeding density increased in the study period from 0.84 to 1.36 calling males per $km^2$. Using the data from the Snēpele route (western Latvia), where counts have been done since the 1960s, and the decrease in proportion of meadows in Latvia, as well as the estimate of Corncrake numbers in Latvia in 1996, I calculated Corncrake numbers in Latvia in the past. The estimate from this model is consistent with previous estimates, *e.g.* for 1995, 20 118 v 22 735 and for 1989-1994, 3726-10 432 v 3 000-10 000. The results suggest that previous warnings that 3 000-10 000 was an underestimate, are incorrect. I now assume that the Corncrake population reached its minimum during the late 1980s and early 1990s and is currently increasing, but is still lower than it was at the beginning of the 20th century. Statistically significant negative correlation ($r=-0.48$; $P=0.045$) was found between total pesticide use in Latvia in year t and Corncrake numbers in the Snēpele route in the following year (t+1). Lower pesticide use indicates reduced income from crop production and thus a lower-intensity (ie nature-friendly) agriculture. Abandonment and low-intensity use of agricultural lands in the1990s are the main reasons for the increase of Corncrake population, but favourable habitat will be lost in the near future by afforestation and intensification of agriculture. My final conclusion is that the Corncrake population is increasing not because of conservation actions (there are none) or legal protection, but because of the crisis in agriculture after the end of the soviet occupation of Latvia. Thus the increase in Corncrake numbers may not persist.

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## 1. Introduction

The Corncrake *Crex crex* has declined in numbers throughout its range since the late 19th century (Tomiałojć 1994) due to intensification of agriculture, and is classified as vulnerable by IUCN (Collar *et al*. 1994). The species has been included in the Red Data Book of Latvia since its establishment in 1980 (Andrušaitis 1985, Keišs 2000) and therefore has been fully protected by law for more than 20 years. Transehe (1965) first indicated a decline in Corncrake numbers in Latvia dating back to World War I. Noticeable declines of the population in Latvia were observed in the 1980s (Priednieks *et al.* 1989), and the first surveys were conducted at several sites at that time (Priednieks *et al.* 1989, Keišs & Ķemlers 2000). Volunteer-based countrywide monitoring of Corncrakes in Latvia started in 1989 (Keišs in press) and still continues. This article analyses causes for the recent increase in Corncrake numbers and speculates about population dynamics of the species in the 20th century based on monitoring data and changes of agricultural land use in Latvia.

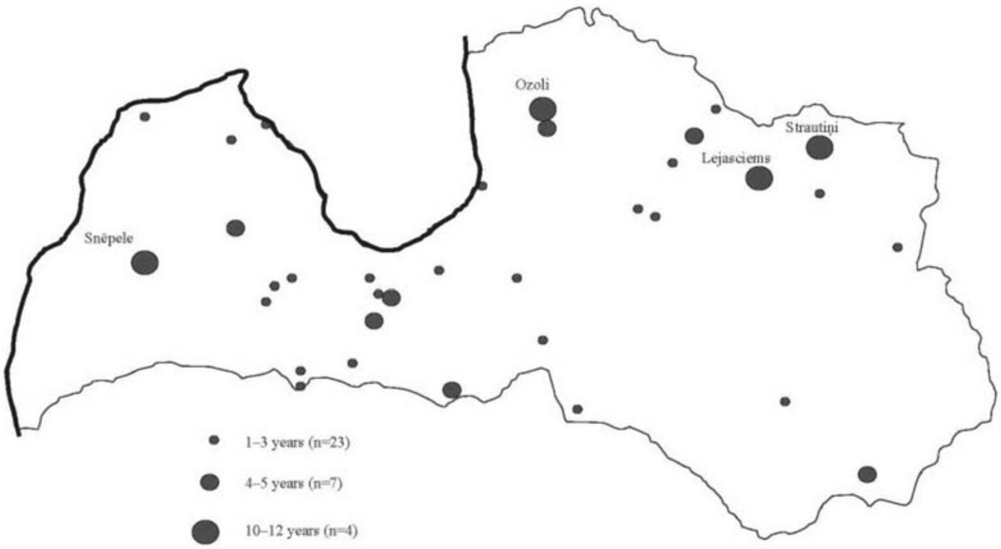


Fig. 1. Locations of sample plots and years the counts have been carried out in each plot.

## 2. Study area and methods

Latvia is located on the coast of the Baltic Sea, in the western part of the East European Plain (highest point, 311m asl). The land covers 64 600km². Forests, peat bogs and scrublands cover 50% of the surface area, and agricultural land 38.5% (Latvian State Land Service 2000).

Corncrake surveys were carried out at 34 survey routes across the country (Fig. 1) during the breeding seasons of 1989-2000. Locations were chosen by volunteer observers and are not randomly distributed. Each surveys route was covered for an average of 3.4 years (1-12, Fig. 1). The number of routes counted in a year varied between 4 and 17 (average 9.7). Routes usually followed countryside roads and observers walked or used a bicycle while counting calling Corncrakes between 2300 and 0300 local time. Nights with frost, rain or strong winds were not recommended for surveys (observers were asked to report local weather conditions before and after of each survey). Two counts per season were recommended, but in 27.6 % of all cases, only one count took place. It was strongly recommended that surveys be carried out before any grass mowing occurred in the area (*e.g.* limit counts to the month of June) and that at least a week should elapse between the counts in a route. Habitat types of calling Corncrakes were determined (observers were asked to map habitats during the day), but habitat data will not be analyzed in this article.

The maximum number of calling Corncrakes per count was used as the annual estimate of the number of breeding pairs in a route. The area of suitable habitats for Corncrakes (all open habitats except for water and villages) covered by each route was estimated by using 1:50 000 topographic map, assuming that Corncrakes might be heard at distances of up to 1km. The area of each route obtained in this way varied between 0.82 and 28.57km² (average 9.32km²).

Information about agriculture in Latvia was obtained from various literature sources (Latvian Agricultural Consultation and Education Centre 1999, Latvian State Land Service 2000, Latvian State Statistical Committee 1991), as well as from unpublished sources of the Faculty of Agronomy, Latvia University of Agriculture. Basic statistical procedures were followed to analyse the data (Liepa 1974).

## 3. Results

During the study period of 12 years, a total of 2498 Corncrakes were registered during all counts. Average density for the total survey period across the routes was 1.81 calling males/km$^2$ (0.29-5.33, SD=1.47, n=33). One route with an average of 11.77 males/km$^2$ across the years was excluded from this analysis, because it has to be excluded statistically by Dixon's criteria (Liepa 1974). Average overall annual density shows an increase over years (r=0.57; *P*=0.0518, Fig. 2), but it fluctuated greatly between 0.67 and 1.90 (x=1.11; SD=0.41; n=12). When we compare the first half of the study period (1989-1994) with the last six years (1995-2000) it is clear that average breeding density has increased (t-test, Fig. 3). This pattern is the same and statistically significant (*P*<0.05) for all pairwise comparisons, if minimum numbers instead of maximum numbers are used for these calculations. A closer look at the four routes counted every year for 10-12 years (for locations, see Fig. 1) shows that Corncrake numbers have significantly increased in Snēpele (Pearson's r=0.77, *P*=0.0034, n=12, area of plot=28.57km$^2$) and Strautiņi (Pearson's r=0.60, *P*=0.0672, n=10, area of plot=6.76km$^2$), have stayed stationary in Lejasciems (Pearson's r=0.24, *P*=0.4830, n=11, area of plot=11.14km$^2$) and have decreased significantly in Ozoli (Pearson's r=-0.89, *P*=0.0002, n=11, area of plot=4.97km$^2$).

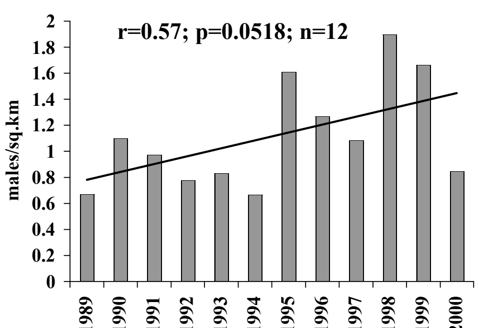


Fig. 2. Dynamics of average Corncrake density in Latvia 1989-2000.

In the early 1960s pesticides (herbicides, fungicides and insecticides) were applied to 184 000ha of agricultural lands in Latvia, this area reaching 1 334 000ha in 1990 (Latvian State Statistical Committee 1991), but fell back dramatically after 1991. In 1994 pesticides were applied to 137 000ha, an area less than for 1960 (Latvian Agricultural Consultation and Education Centre 1999). The number of Corncrakes observed in the Snēpele sample plot has a significant negative correlation with the total pesticide use in Latvia in the previous year (Pearson's r=-0.48, *P*=0.045, n=17, Fig. 4). In this analysis, additional observations in Snēpele since the 1960s were also included (Keišs

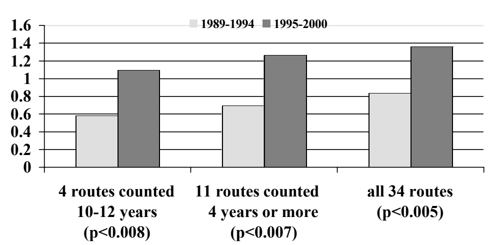


Fig. 3. Corncrake breeding density increases in Latvia in 1995-2000 v 1989-1994 (t-test).

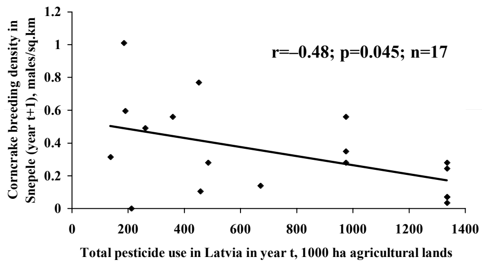


Fig. 4. Pearson's correlation between total pesticide use (area, x10$^3$ha agricultural lands) in Latvia (in year t) with Corncrake breeding density (males/km$^2$) in Snēpele plot in the following year (year t+1).

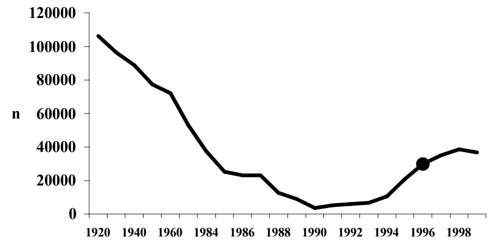

Fig. 5. Calculated dynamics of Corncrake population in Latvia in the 20$^{th}$ century (based on the estimate of 38 000 calling males in 1996).

& Ķemlers 2000). The proportion of meadows in agricultural lands in Latvia had already decreased by almost half before 1960 (1910, 31%; 1960, 17%) and continued to decrease to 13% in 1990 (P. Šķiņķis unpubl).

A simplified model to back-calculate numbers of Corncrakes was developed assuming that:

1. The number of Corncrakes in Latvia is directly proportional to the area of meadows in Latvia.
2. The intensity of meadow use has increased gradually during the past century.
3. The dynamics of Corncrakes in Latvia have been the same as in Snēpele from the 1960s to 1980s.
4. The maximum estimate of numbers in the 1996 survey (38 000, Keišs 1997) is the most likely number of actual population size of Corncrakes in Latvia in 1996.

## 4. Discussion

Despite the low number of continuous monitoring routes, I conclude that the number of Corncrakes has increased in Latvia during the last decade. Since there has been no change in formal or actual protection policies of this species in Latvia, this increase cannot be associated with any conservation efforts.

There are no quantitative data on Corncrakes in Latvia prior to the 1960s and the 1960s data are anecdotal (Keišs & Ķemlers 2000). Therefore, modelling is the only way to get even rough approximations of how large the Corncrake population could have been in the past. My model is very superficial (Fig. 5), but it matches fairly well with previous attempts of population estimates in the 1990s. Strazds *et al*. (1994) estimated Latvian Corncrake population at 3000-10 000 calling males; the model gives 3726-10 432 for the years of 1989-1994. Keišs (in press) estimated the Corncrake population in 1995 to be ≥22 000; the model for 1995 gives 20 118. Therefore, I conclude that my previously expressed concerns that estimate of 3000-10 000 are an underestimate (Keišs in press), are not valid. It might be true that Corncrake numbers were at an all time low in Latvia in the late 1980s and early 1990s.

In 1996, meadows held 42.75% of the Corncrake population in Latvia (Keišs 1997). Therefore, meadows are the most important habitat for Corncrakes. The proportion of meadows in agricultural lands in Latvia had already decreased by almost half by the 1960s, in comparison with the early 20th century (17% v 31%). Therefore it appears that Corncrake numbers had already reduced markedly by the 1960s. In fact, a decline of Corncrake numbers at the beginning of the 20th century is noted by Transehe (1965). The total area of agricultural lands between 1940 and 1990 decreased by 11 466 000ha (31%) and the area of meadows by 670 000ha (73%). Undoubtedly in the first years after abandonment, these areas were favoured by Corncrakes, but subsequent encroachment by bushes makes the land unsuitable. Since there was no disturbance by agriculture, I assume the nesting success was high in the early years, and if we assume that land abandonment during the soviet era (1940-1990) was gradual, it might have contributed to a slower decrease of Corncrakes in Eastern European countries.

Recovery of the Corncrake population in Latvia in the recent years is directly related to the crisis in agriculture. Use of pesticides has decreased almost 10 times (applied on 1 333 800ha in 1990 v 136 700ha in 1994), which might have benefited the Corncrake. In spring crops, when most pesticides (specifically herbicides) are applied, their effects can still limit Corncrake numbers (Ā. Leilands pers comm), but I presume that pesticides are more important as an indicator of agricultural intensity: the less pesticides are used, the less intensive are other agricultural practices. I suspect that, for example, the large-scale agricultural 'melioration' of wet meadows (64% of all agricultural lands in Latvia were 'meliorated' by 1995) had a much higher negative effect on the Corncrake population. Natural processes might have been allowed to take place in part of the land: 11.1% of all agricultural lands were abandoned in 1995, increasing to 17.5% in 1999. Corncrakes quickly took advantage of this situation: 28.85% of the population in 1996 lived on abandoned agricultural lands, making this habitat the second most important in Latvia (Keišs 1997). However, this is not the long-term situation because12 600ha had been overgrown by shrubs by 1995, rising by 1999 to 26 500ha, and these are likely to be underestimates. The plans for agricultural land use in Latvia include afforestation of as much as 10% in the near future. The remaining agricultural land is expected to be used more intensively, approaching West European standards, to become competitively profitable. Both the afforestation and agricultural intensification will have negative effects on the Corncrake. In addition, change of land use from meadows to arable lands was observed in Latvia in the 1990s (see also Auniņš & Priednieks, 2003) and it is expected to continue. The observed severe declines in the Ozoli sample plot in the last decade are attributed purely to changes in land use from winter crops (Corncrake habitat) to potato fields (no habitat for Corncrakes).

Many places in Latvia with historical low-intensity meadows have been abandoned for more than a decade already. Several of them have been recently designated as Important Bird Areas for Corncrakes (Račinskis & Stípniece 2000). Finding ways of keeping these areas from turning into scrublands would be one of the most important nature conservation goals in Latvia. Some of these areas do not have significant amount of shrubs even 30 years after agricultural activities ceased (Víksne 1997). Additional research on soils and vegetation could possibly help answer the question whether low cost maintenance of semi-natural meadows is possible as proposed by Flade (1997).

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**Ornis Hungarica 12-13: 157-167. 2003**

# The census and distribution of wintering woodpigeons *Columba palumbus* in the Iberian peninsula

A. Bea, R. Beitia and J. M. Fernández

Bea, A., Beitia, R. and Fernández, J. M. 2003. The census and distribution of wintering woodpigeons *Columba palumbus* in the Iberian peninsula. – Ornis Hung. 12-13: 157-167.

Throughout the 1997-1998, 1998-1999 and 1999-2000 seasons, Woodpigeon population censuses were carried out in their traditional Iberian wintering area, which comprises the southwestern quadrant of the Peninsula. The method applied was the direct counting of flocks in communal roosts, of which 210 have been checked so far. Four counts were carried out per season in November, December, January and February. As an average, 95% of the roosts were counted simultaneously. Between 140 and 230 collaborators participated in each one of the censuses. These collaborators were mainly wardens from the Spanish autonomous regions and the Portuguese Government. Quantitative data suggests that the wintering population in the study area would total about 2.5-3 million birds, not including inter-yearly oscillations associated with reproductive success in the breeding areas. Their gregariousness remained relatively constant between November and January, 40-70% of the birds being counted in roosts of more than 400 000 individuals (1-2% of the number of roosts). Numbers were stable in this period, in spite of some inter-monthly differences attributed to difficulties in obtaining full cover. However, February offered a variable pattern. This phenomenon shows that dates of start of the spring migration must vary from year to year. The distribution of these contingents was not homogeneous within the study area, a massive presence occurring in Portugal and Extremadura-Toledo alternately. The evaluation of acorn production in Iberian *dehesas* and *montados* (pastureland) during 1999 provided interesting points for interpretation. For example, the low availability of food in cork tree woods during that winter explained the absence of birds in the Portuguese districts, and may have stimulated the start of migratory movements. The seemingly direct response shown by wintering Woodpigeons toward the nutritional capacity of the *dehesa* suggests that the analysis of this factor could allow predictions of the spatial and seasonal distribution of the population.

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## 1. Introduction

The Woodpigeon *Columba palumbus* is a member of the family Columbidae, a Palearctic, though mainly European, species. The subspecies *C.p. palumbus* breeds across the continent, from Russia and Scandinavia to the Atlantic coasts and Mediterranean peninsulas (Saari 1997). Previously regarded as a woodland bird, its recent adaptation to, and trophic dependence upon cultivated fields has led to numerical prosperity in several European countries, allowing phenologic, behavioural and demographic changes to be

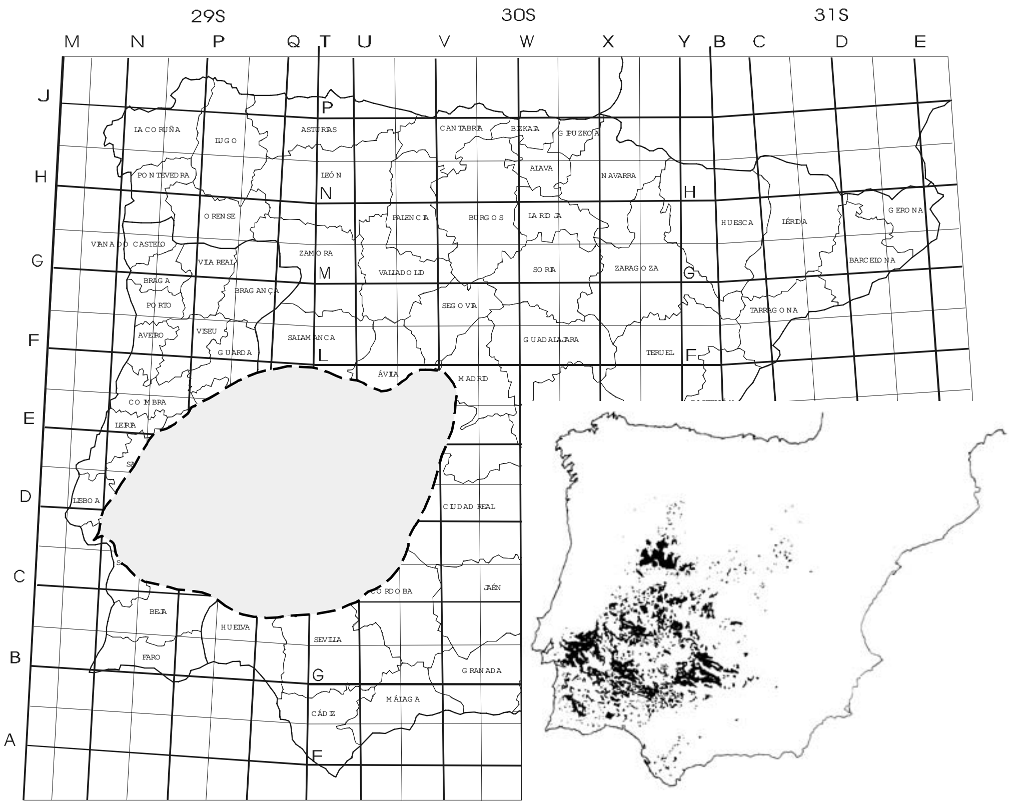


Fig. 1. The study area in the Iberian Peninsula and the range of dehesas and montados.

explained through shifts in agricultural techniques (O'Connor & Shrubb 1986, Inglis *et al.* 1990).

Recognised migration patterns include partial or basically migratory populations, such as Fennoscandian and eastern European, as well as short-distance migratory and resident populations, as in Western Europe. This pattern has been described as a general alohiemism (Bernis 1966), because northern and eastern populations occupy winter quarters further south and west than intermediate populations. In this sense, the important role of the Iberian Peninsula as a wintering ground was first pointed out by Bernis (1967) and strengthened by Purroy (1988).

The Woodpigeon is a prominent game species in many European regions as shown both by the total hunting bag across Europe, estimated at 9.5m birds yearly during the 1980s, (Purroy *et al.* 1984) and by such as social and cultural events related to the shooting season, for example Région Cynégétique du Sud-Ouest (1994).

Accurate population size information is a basic requirement for establishing a reliable monitoring program. Tracking this variable (and others such as reproductive success or survival rates) over periods of time will give early indication of population trends, whose appropriate interpretation may suggest managing or conservation measures. Thus, monitoring is a procedure to measure the history of variables in a systematic way, but with explicit aims

(Spellerberg 1991, Gilbert *et al.* 1998). The present study is a contribution to the quantitative knowledge and distribution of Woodpigeon population wintering in the Iberian Peninsula, and it also explores some of the applicable environmental factors.

## 2. Study Area and Methods

The study area approximates to the southwestern quadrant of the Iberian Peninsula and partially includes the Spanish regions of Castilla y León, Castilla-La Mancha, Madrid, Extremadura and Andalucía, and Portuguese Alentejo and Ribatejo (Fig. 1). The quadrant was defined after the analysis of the geographic distribution of recoveries during the winter months of December and January of birds ringed in many European countries, 88% of recoveries being entered in databases up to 1997 (N=58) of Spanish and Portuguese ringing schemes (Dirección General de Conservación de la Naturaleza, Instituto da Conservaçao da Natureza and Sociedad de Ciencias Aranzadi) were from inside the quadrant. Bernis (1967) had used the same method to define the wintering quarters of trans-Pyrenean migratory Woodpigeons.

The area of the quadrant corresponds largely to those regions whose landscape mostly comprises *dehesas* and *montados*. These habitat types are principally wooded pasturelands with holm oaks *Quercus ilex* and cork oaks *Quercus suber* in densities of 20-60 trees/ha, and have multifarious productive uses, cattle raising in holm oak woodland and cork exploitation in cork oak woodland being relatively important (Gómez 1997). The total extent of these habitats in the Iberian Peninsula depends on the defining criteria, but *c*3.1m hectares is a reasonable estimate (Díaz *et al.* 1997).

Of course, other areas in Iberia hold wintering Woodpigeons, and these have also been studied. However, winter densities there are much lower than in spring and summer (Díaz *et al.* 1996), and it is possible that wintering birds belong to the more sedentary native Iberian populations. That some from these populations move towards the southwestern quadrant (Gallego 1985) would explain the density reductions. Lack of ringing effort applied to resident populations obscures knowledge about their migration patterns.

The method employed to undertake a census of wintering Woodpigeon populations in the study area was adapted from that used in France for the same purpose (Région Cynégétique du Sud-Ouest, 1994). The first phase, an inventory of roosting sites, was carried out in 1997, and was achieved through personal inquiries to administration officials, wardens, shooting federations, birdwatchers, naturalists, local people and others involved in wildlife management. A database was compiled of ‘sites’ mentioned by any informant. A ‘site’ was defined as any location noted ever to have been used as roosts by Woodpigeons. The database was constantly updated to delete ‘sites’ that had been felled or burnt down and to include newly discovered roosts. At the time of writing, the total set comprises 218 roosts (including former ‘sites’), 118 of them being in Spain and 100 in Portugal.

It was not possible to use the data from inquiries to make rough estimates of population sizes and trends, because many observations were inaccurate and roosting behaviour was not consistent at any particular site.

The population census method chosen was to count directly the birds gathered in known roosting sites, a technique applied generally to censuses of gregarious wintering birds such as herons, gulls and starlings, because it allows absolute numbers to be established by taking into account the contiguous distribution of birds and separating the counts into detectable recognisable units (Tellería 1986). An important requirement is for counts to be simultaneous, so that possible biases, due to bird mobility, dispersion or interchange between roosts, are avoided. Because of the high number of roosts to be visited, 140 to 230 observers (mainly official and wardens) collaborated in each census period, so that an average of 95% of counts were done simultaneously on the planned dates.

Censuses were performed in the winters of 1997-1998, 1998-1999 and 1999-2000. In each winter four census periods were established, in November, December, January and February (except 1997-98, when there was no February count). Dawn counts are preferable, because the movement to the feeding areas is direct and occurs in large flocks at sunrise and over a short timescale. Dusk counts occur over a longer period, but because the birds often use pre-roosting sites and are unsettled, the risk of double-counting is high. In any case, the census methodology included visiting the roosting sites the previous evening, when the arrival headings to roost usually were but one or two, the next morning's departure headings being the exact opposite. Knowing these routines helped the choice of the most favourable observation points. Observers made prudent rough estimatesof numbers at dusk in case the weather prevented proper counts the next morning.

Observers received training sessions to explain study goals, the species' natural history, census procedure and the methods to be employed to count large flocks (divide them into lesser units; Bibby *et al.* 1993). Great emphasis was placed on these sessions in order to stimulate responsible and effective participation and to reduce bias due to unsound counts. Good observer training is one of the factors that increases counting accuracy (Erwin 1982, Cantos & Tellería 1985).

As well as this net of collaborators, we formed a mobile team of experienced observers, its task being to detect and count bigger roosts whose size exceeded normal counting skills. Alternative techniques were used, such as counting different flocks of birds by different groups of observers, and taking advantage of boundaries as marking points while flocks passed them.

During the overall counting period, we began in 1999 an evaluation of acorn production in Iberian *dehesas* and *montados*. We selected at random 25 routes on secondary and country roads within the study area (18 in Spain and 7 in Portugal). In each, 25 sample points were selected, evaluating acorn production in a randomly-selected tree to produce a semi-quantitative index. The average acorn crop in each route or wood came from the expression:

$$P_i=\Sigma(N_p \times I_p)/25$$

and

$$N_p=A_p/[(D_p/100)^{2\times}(\pi/4)]$$

where $P_i$ is the average crop in the

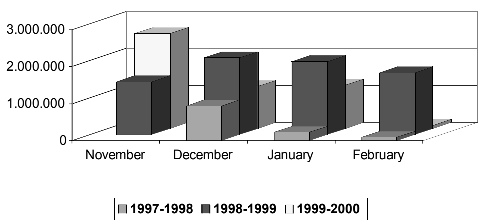


Fig. 2. Results of Woodpigeon censuses in 1997-1998, 1998-1999 and 1999-2000 winters.

route, $N_p$ the equivalent number of trees in the 1ha plot throughout the sample point, $I_p$ the semi-quantitative index of acorn production in the particular tree, $D_p$ its diameter (cm), and $A_p$ the basal area in the plot ($m^2$/ha).

This method, developed by Instituto de la Madera, el Corcho y el Carbón of the Junta de Extremadura, provides a simple and rapid sampling technique to be used in field conditions, and gives accurate measurements (C. Bernal *pers comm*). Samples were taken in the first fortnight of October, when fruit growth allows visibility from the ground, but the state of fruit maturity precludes it from being lost through falling.

## 3. Results

Fig. 2 shows the total numbers of Woodpigeons counted in November, December, January and February for the winters of 1997-1998, 1998-1999 and 1999-2000. The population size is assessed at 2.5 to 3 million birds (disregarding inter-annual fluctuations) It has been assumed that the number of birds using unknown or unvisited roosting sites, or sleeping in dispersed flocks are not significant, although these were not counted. The lower figures for 1997-98 are probably due to observer inexperience or to

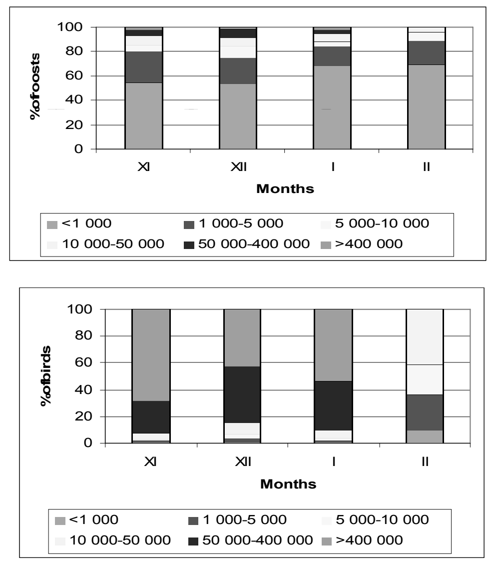


Fig. 3. Percentage of occupied roosts by roost-size category (above) and percentage of birds counted by roost-size category (below).

problems of achieving observer cover or proper coordination in the first winter, rather than to a lack of birds.

The cover achieved of roosting sites (sites planned to be visited/sites actually visited) averaged 88% in 1998-99, declining to 65% in 1999-2000. This reduction in achieved cover could also explain the decline in Woodpigeon numbers between November and December 1999, for it is difficult to attribute to factors other than census performance. The November cover of 70%, declined to 63% in December. Given the widespread gregarious distribution of the birds, the level of cover acts an indicator of counting effort, but not in a way proportional to the censused population, because only a few unvisited sites holding significantly large roosts would result in reductions to the overall totals counted.

Also relevant was the contrasting numerical stability during the winters of 1998-99 and 1999-2000 for the period of

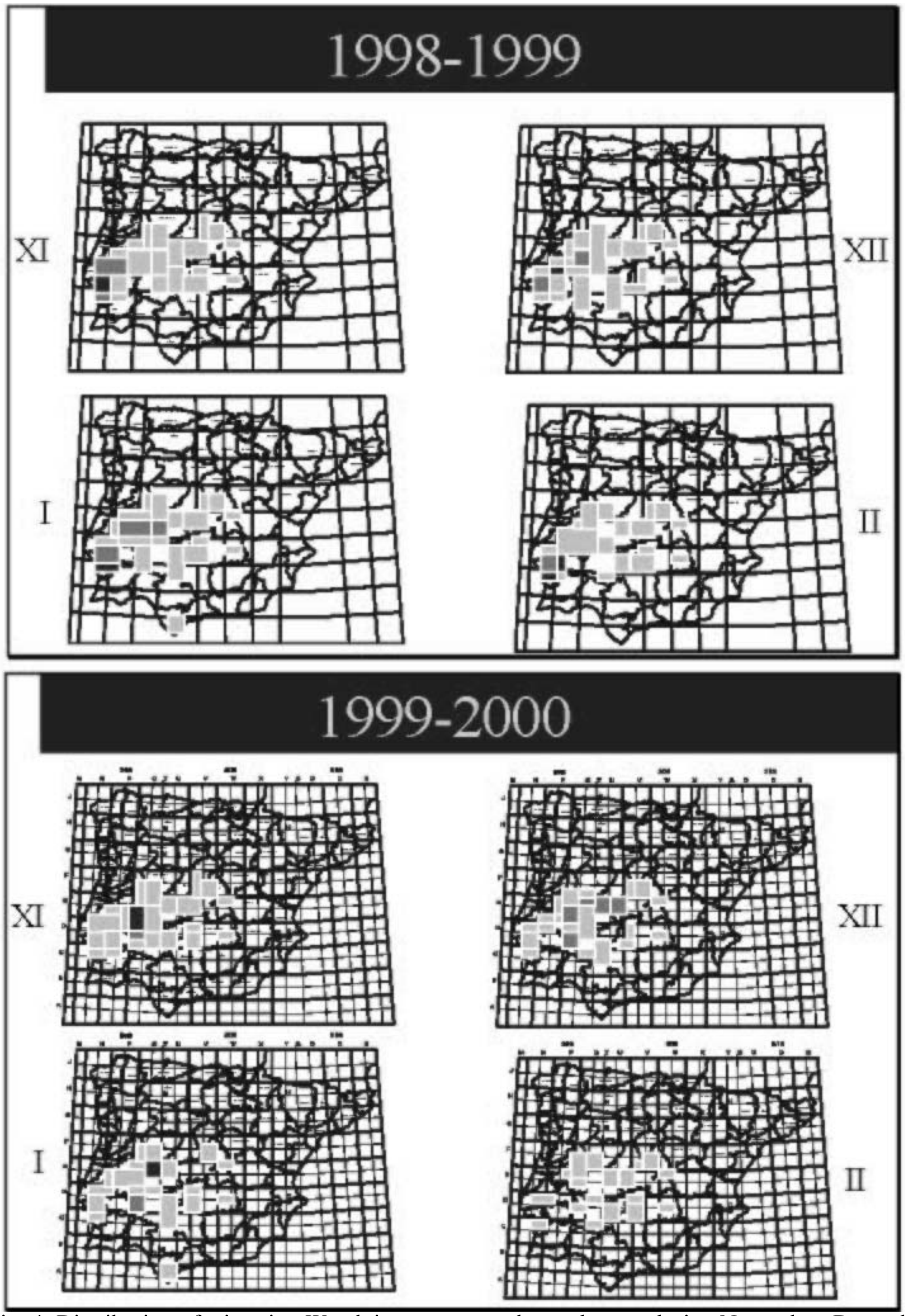


Fig. 4. Distribution of wintering Woodpigeons across the study area during November, December, January and February of 1998-1999 (above) and 1999-2000 (below). Light grey squares stand for fewer than 50 000 birds counted in that particular square; dark grey squares stand for between 50 000 to 500 000 birds; black squares stand for more than 500 000 birds.

January and February. In 1998-99, the numbers reduced by only 15.5%, but in 1999-2000, by 93.1%. This difference cannot be attributed to variations in counting cover, but reflected shifts in gregarious behaviour that either stimulated a reduction in roost size, or initiated early return migration. In the first case, the birds would not have been in the roosts to be counted because of short-range dispersal within the study area, and in the second, the birds would have left the study area entirely.

To describe the gregarious behaviour of Woodpigeons, the roost occupation and population held in winter 1999-2000 were categorized by roost-size (Fig. 3). About 80% of roosts each held fewer than 5000 Woodpigeons, but less than 10% held more than 50 000 birds each. Yet they held 80-90% of the censused population, demonstrating the intense gregariousness of the species. Large aggregations of course are more detectable than small, but it is unlikely that significant numbers of dispersed, smaller aggregations were missed. This pattern clearly was modified by February, as described above.

In Fig. 4, the distribution is shown of Woodpigeon numbers within the study area for 1998-1999. The most remarkable feature during the four count periods was that the bulk of the population remained in Portuguese coastal districts. Fig. 4 also shows that the distribution during the 1999-2000 winter (except for February 2000, as surmised above) was biased towards the interior regions, mainly in Spain

The average acorn crop estimated throughout all the selected routes in 1999 was 367.5kg/ha, a value considered as medium-low against accepted standard ranges by Ceballos & Ruiz de la Torre (1971) and Montoya (1989). Nevertheless, clear variable patterns between cork oak and holm oak acorn production were found. Average crops in cork-oak dominated pasturelands 86.4kg/ha ($\sigma$=43.8, N=5), against 438.2kg/ha ($\sigma$=233,4, N=20) in holm-oak dominated pasturelands, the difference being highly significant (t=3.3, $P$<0.01). Furthermore, crop variability in cork oaks was low, with poor production everywhere (apparently synchronized), but results from holm oaks pasturelands showed greater dispersion. Also of interest was that the acorn crop in Spanish *dehesas* and Portuguese *montados* (independent of tree species) differed significantly (457.9 kg/ha against 136.1 kg/ha; t=3.44, $P$<0.01).

## 4. Discussion

Through evaluation of the acorn crop in Iberian *dehesas* and *montados*, the distribution of Woodpigeon population in the study area can be explained satisfactorily. Acorns from cork and holm oaks are a principle winter food for this population, as are dicotyledonous leaves at the beginning of the season (Purroy *et al.* 1984). However, cork and holm oaks show a complementary distribution in the Iberian Peninsula as a consequence of different ecological preferences. Cork oaks occupy sectors possessing mild oceanic climate, relatively high humidity and a low incidence of frost. Holm oaks are more resistant to cold and temperature contrasts, so that their basic distribution lies across sectors of continental climate (Gómez 1997). In this way, the abundance of cork oak pasturelands in the landscape increases from NE to SW, being dominant in Portuguese districts, nearly exclusively so coastally. In the 1999-2000 winter, the study area lacked a cork oak crop, and so the Woodpigeon distribution shifted to regions where holm oak pasturelands were dominant, mainly in Spain. Crop levels measured in 1999 in *dehesas* and *montados* were confirmed by other independent evaluation schemes (C. Bernal *pers comm,* Vázquez *et al.* 2000).

This close fit between the spatial distribution of Woodpigeons and their *potential* trophic resources (a provisional hypothesis that will require data series from several years) is observed at wider, landscape scale. Locally, the actual food availability is influenced by some extremely variable factors (Borchert *et al.* 1989). Woodpigeon food shortages may arise from an abundance of seed predators, whether invertebrates (beetles and caterpillars, [Vázquez 1998]) or vertebrates (cattle, and to a lesser extent, rodents), and their level of intake, which in turn is related to the existence of alternative resources for the seed predators (Pulido 1999).

In man-made ecosystems such as *dehesas* and *montados*, management can determine the productivity of the trees, by external means (farming and forestry practices) and by exploitation of internal (genetic) variations (Koenig *et al.* 1991), thus determining fruit availability. For instance, the degree of shrub cover in *dehesas* was positively related to rodent abundance (Díaz *et al.* 1993). Moreover, shrub regeneration depends on grazing pressure and the economic orientation of each individual *dehesa* (San Miguel 1999).

In our samples, cork oak pasturelands showed greater basal area, tree density and shrub cover than holm oak pastureland, although the differences were statistically significant only in the first and second parameters (t test, $P<0,01$). These structural features could play a role at a detailed scale in the way Woodpigeons select feeding areas, but there are no real data on that topic. Basal area and tree density (closely and positively related; $r=0.8$, $t=6.2$, $P<0.001$) influence production, and shrub cover would favour predation pressure on Woodpigeons and an abundance of seed-eating rodents.

As a whole, even though at a detailed scale the above-mentioned factors are responsible for great variability between plots, the close fit between the patterns of distribution of potential food resources and of the birds is circumstantial evidence that such resources are limiting environmental factors (Newton 1998). In Great Britain and Sweden, the effect of food availability on Woodpigeon densities in the post-breeding period repeatedly has been shown (Murton *et al.* 1964, Nilsson 1984, Inglis *et al.* 1990). In the Iberian Peninsula, crop variability in holm oak pasturelands would allow the maintenance of body fitness and weight in wintering birds (Purroy *et al.* 1984) in spite of the greater reproductive synchronism of cork oak pasturelands. Both types of habitat could be described as having almost complementary roles, which probably allowed the Iberian Woodpigeon winter population to evolve as a dynamically stable component of the species' migration strategy.

We think it would be unreliable for population trends to be drawn from population figures from the present direct census and from the previous estimates obtained since the early 1980s (5-6 million wintering birds [Purroy & Rodero 1986]), because the respective methodologies were substantially different. Since 1981, direct, season-long counts of migrating flocks of Woodpigeons have been made at the Iraty pass in the French Pyrenees (Région Cynégétique du Sud-Ouest 1994). Trends show this migratory population to be declining slightly, for reasons that are unclear; there may be a real decrease, or the migration route may have changed.

Nevertheless, the range of the wintering population has probably reduced, because the presence of birds in southern *dehesas* was not verified in the 1997-2000 period, despite good food availability. Twenty years previously, the use of this area was evident, at least in some winter months (Purroy *et al.* 1984). Fidelity to winter ranges could play a role in the regular occupancy of Tajo-Sado basin, between the Toledo and Setúbal.

There is some evidences that the start of the return (spring) migration may change between years, for in February 2000 there were several observations and reports of flocks migrating through the central Spanish mountains. It would appear that these movements flout the hypothesis of dispersion of flocking birds, and as yet there is no explanation. Complex mechanisms, genetically fixed and of an hormonal nature, act to stimulate migratory behaviour in birds as *proximate* factors (Berthold 1993). We note that the modulating or synchronizing effects of some environmental conditions have been proved, especially for species that have a northerly breeding range, such as for Common Crane *Grus grus* (Alonso *et al.* 1990).

We suggest that food availability in the Woodpigeons Iberian wintering range influences the start of return migration. The reproductive strategies of cork and holm oaks shows differing temporal patterns, the former having a sustained crop for a longer period through the winter months (Cańellas *et al.* 1991, Cańellas 1993). Consequently, we suggest that the holm oak pasturelands crop would be more likely to be consumed by seed predators, especially when production is medium or low, as has been observed in *Fagaceae* trees in temperate latitudes (Crawley 1992, Siscart *et al.* 1999) and particularly in *dehesas* of holm oaks (Pulido 1999) and cork oaks (Herrera 1995, Vázquez *et al.* 1997). Reproductive strategies and plant recruitment in these species is generally interpreted under the "satiation of consumers hypothesis" (Janzen, 1971), which is verifiable in highly-productive fruit seasons that exceed the consumption capacity of crop predators.

Finally, the described fit between Woodpigeon distribution and the species' potential food resources, even with predicted capacity at meso-scale or regional scale (Newton, 1998), would relegate to secondary roles other factors sometimes claimed to explain so-called spatio-temporal "irregular" wintering. Such factors include shooting or the degree of protection afforded by roosting sites against meteorological conditions (Purroy *et al.* 1984), although these could have a local influence over habitat selection in relation to feeding or roosting areas.

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Geral das Florestas and from shooting federations in southwestern France. The regions of Aquitaine, Midi-Pyrénées and País Vasco also have to be thanked. Funds were provided by Fondo de Cooperación Euskadi-Aquitania, Union des Federations Departementales des Chasseurs and Federación Espańola de Caza.

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**Ornis Hungarica 12-13: 169-182. 2003**

# Integrated population monitoring of sand martin *Riparia riparia* - an opportunity to monitor the effects of environmental disasters along the river Tisza

T. Szép, Z. Szabó D. and J. Vallner

Szép, T., Z. Szabó D. and Vallner, J. 2003. Integrated population monitoring of sand martin *Riparia riparia* - an opportunity to monitor the effects of environmental disasters along the river Tisza – Ornis Hung. 12-13: 169-182.

From January to March 2000, the entire length of the River Tisza suffered an appallingly serious environmental disaster when the collapse of tailings dams belonging to upriver Romanian gold mines caused severe pollution by cyanide and heavy metals. This pollution was the direct cause the death of flora and fauna in the Tisza along its length and threatened the entire ecosystem of the river, one of the last remaining natural major rivers in Central Europe almost free of large-scale man-made developments. The river Tisza is such an important breeding and roosting area for large populations of several insectivorous and piscivorous bird species that several Important Bird Areas (IBAs) have been established along its course. Most of these species are migrants that fortuitously happened to be elsewhere in their travels when the pollution occurred, but the scale of the problem was such that delayed impacts can be expected. As it happens, long-term integrated monitoring work on the Sand Martin *Riparia riparia* breeding population along the river Tisza in Hungary has been running since 1986 under the aegis of MME and BirdLife Hungary. This project also happens to monitor the population size and distribution of Kingfisher *Alcedo atthis* and so was well placed to begin comprehensive monitoring of the short-, mid- and long-term effects of this disaster. The breeding populations of these two species along the river Tisza depend predominantly on the supply of their food, the fauna of the river and its flood zone. The two species, by macabre good fortune, happen to be the ideal models for studying the effects of the disaster on insectivorous and piscivorous birds. Detailed studies in 2000 following proven protocols, such as fieldwork and chemical analysis of the feathers, revealed that the pollution has had no measurable effects on population sizes, distribution and reproductive success, and that the level of heavy metals in the food chain of insectivorous birds did not increase. However, precedent and the scale of the disaster suggest that the lack of immediate effects means that there may well be secondary effects in the longer term from subsequent events, such as floods and droughts. The disaster has brought greater international awareness, which may help to reduce pollution or make such incidents less likely. Our investigation underlined the importance of monitoring in these kinds of habitats, because it showed that some assumptions about the consequences of the accident were wrong; in the absence of data, there is a risk in such circumstances of misinterpreting the outcome.

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## 1. Introduction

From January to March 2000, the length of the River Tisza suffered an appallingly serious environmental disaster when the collapse of tailings dams belonging to upriver Romanian gold mines (Anon 2000) caused severe pollution by cyanide and heavy metals. The pollution was the direct cause of widespread killing of flora and fauna in the Tisza and it threatened the entire ecosystem of the river, one of the last natural major rivers in Central Europe that remains almost free of large-scale man-made developments.

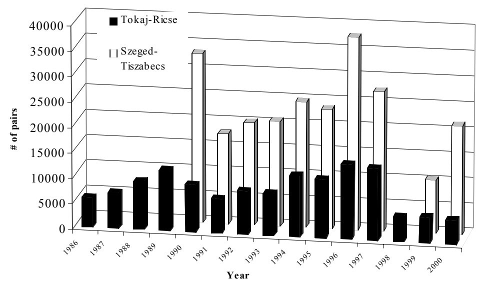


Fig. 1. Numbers of breeding Sand Martin (pairs) along the Hungarian section of the river Tisza (Szeged-Tiszabecs) and along the upper part of the Tisza (Tokaj-Ricse). In 1998 (in the upper section of the Tisza, the population size was surveyed one week before the flood in June that later destroyed these holes.

The River Tisza is a very important breeding and roosting area for large populations of several insectivorous and piscivorous bird species. Along its course lie several IBAs (Heath & Evans 2000, Nagy 1998). Most of these species are migratory birds that happened to be elsewhere in their travels when the pollution occurred, but the scale of the problem meant that delayed impacts could be expected as was the case at Dońana (Meharg *et al*. 1999).

However, considering the scale and extent of the pollution, we could not exclude the possibility of several direct and indirect mechanisms affecting the populations arriving and breeding along

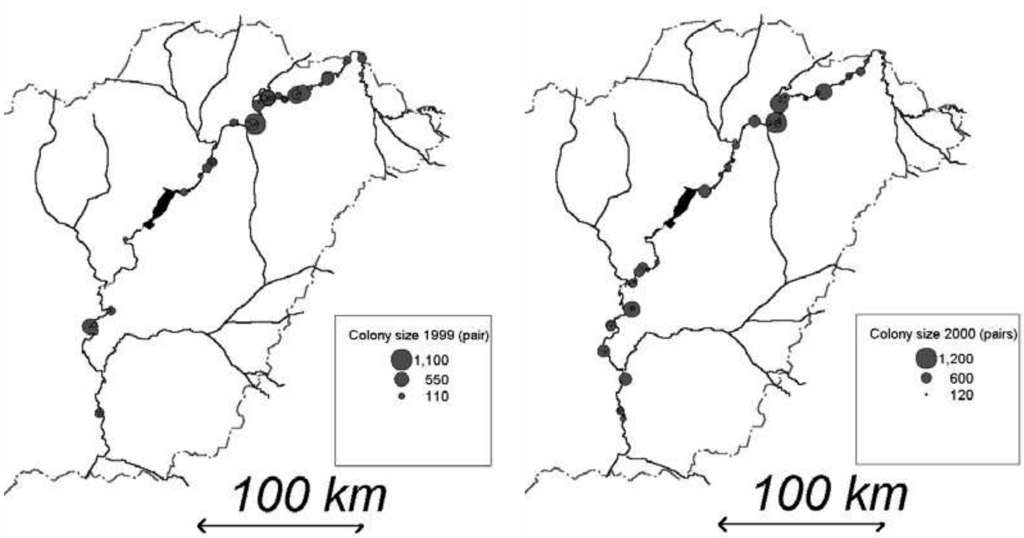


Fig. 2. Distribution and size of Sand Martin colonies along the Hungarian section of the river Tisza in 1999 (left) and 2000 (right).

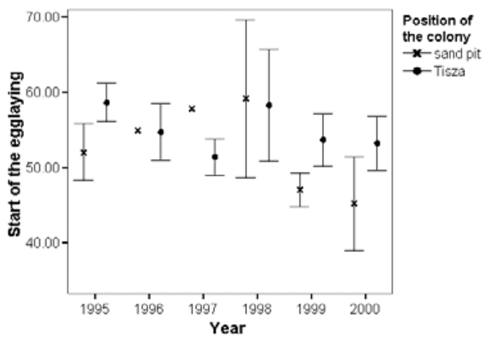


Fig. 3. Average start of egg-laying in colonies along the river Tisza and in sandpits remote from the river (with SE). Day 1 refers to 01 April. In 1996 and 1997 we include data from only one sandpit colony.

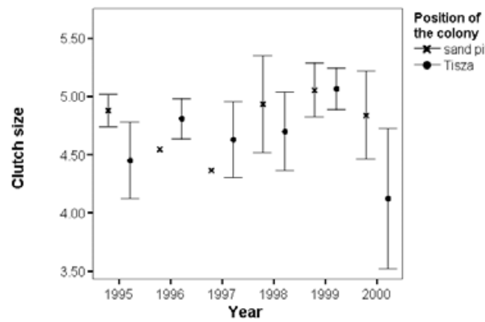


Fig. 4. Average size of the first clutch in colonies along the river Tisza and in sandpits far remote the river (with SE). In 1996 and 1997 we include data from only one sandpit colony.

the river several months after the disaster. Examples are:

1. Population decreases because drastic changes in the quality of the breeding habitat perturbed the food supply.
2. Negative effects on reproduction because of the toxic effects of heavy metal pollution and through shortage of food.

In this highly natural system, it was also important to consider natural events that could have large impacts on the flora and fauna. In April and May 2000, two months after the pollution struck, very severe floods occurred throughout the Tisza system, higher than any recorded since the 1850s, and these severely impacted on breeding bird success.

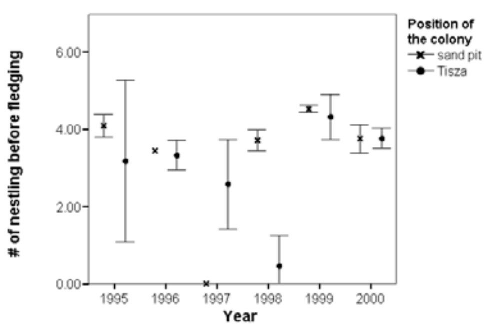


Fig. 5. Average numbers of nestling before fledging (14-18 days old) in colonies along the river Tisza and in sandpits remote from the river (with SE). In 1996 we include data from only one sandpit colony; in 1997 there was no sandpit colony with fledged nestlings.

Long-term integrated monitoring work on the Sand Martin breeding population along the river Tisza (Szép 1991a) in Hungary has been running since 1986 under the aegis of MME and now BirdLife Hungary (Szép 1991b, 1995). This project provided the unwanted but valuable opportunity not only to monitor how the disaster affected the Sand Martin and the population size and distribution of Kingfisher but also to monitor the disaster's short-, mid- and long-term consequences. There would, in addition, be the chance of analysing the impacts of other natural and man-made effects on both insectivorous and piscivorous species (Furness & Greenwood 1993).

The breeding populations of these two species along the river Tisza depend predominantly on their food supply that originates mainly from the river fauna and the associated flooded areas. On the basis of past investigations of the foraging behaviour of breeding Sand Martin (Alves & Johnstone 1994) we could expect that the pattern of the parents collecting a large percentage of their food within a one-kilometre radius would also exist for the Kingfisher (Cramp 1990). Sand Martin and Kingfisher populations along the river could fulfil the criteria for indicator species for insectivorous and piscivorous birds respectively, because:

1. A large percentage of their food is collected above or in the river and its flood area.
2. Both species exhibit a strong dependence on the food supply associated with the river.

In the case of Sand Martin, there are methods and studies available to check the potential effects of food supply on reproduction (Szép & Moller 1999). Furthermore, methods of analysis of the chemical composition of feathers have been developed, techniques that could provide relevant information about the accumulation of heavy metals (Vallner *et al.* 1999, 2000).

The aim of this paper is to present:

1. How a long-term integrated monitoring study on breeding bird species could provide information about the effects of a drastic environmental pollution incident.
2. The importance of detailed investigations for the identification the effects of natural and man-made events on the ecosystem of a natural river.

## 2. Study area and methods

### 2.1 Surveying nesting habitats and breeding populations

In the framework of integrated monitoring of the Sand Martin population, we surveyed all potential breeding walls (whose perpendicular height forming part of the bank is >0.5 m) in three areas:

1. Along the Hungarian section of the river Tisza between Tiszabecs and Szeged (*c*580 km) once per year (since

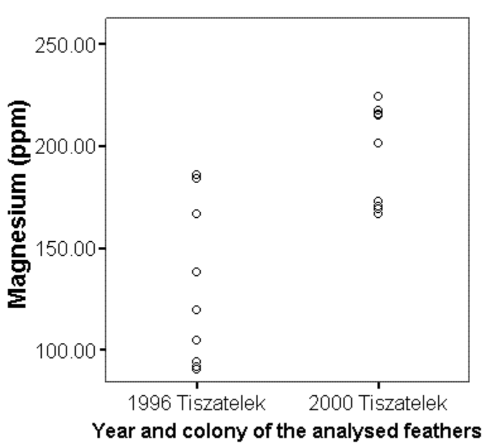


Fig. 6. Concentrations of magnesium (ppm) in the tail feathers of juvenile Sand Martin hatched at Tiszatelek colony along the river Tisza in 1996 and 2000.

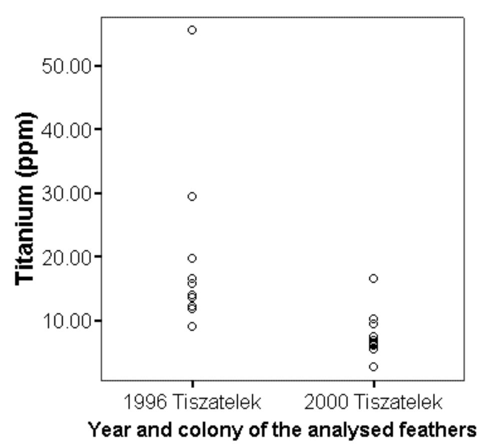


Fig. 7. Concentrations of titanium (ppm) in the tail feathers of juvenile Sand Martin hatched at Tiszatelek colony along the river Tisza in 1996 and 2000.

1990) from a motorboat during July and August.

2. Along the upper section of the river Tisza between Tuzsér and Tokaj (72 km) twice per week (since 1986) between mid-April and mid-August.
3. In sand pits located in a 20 km-wide belt along the studied upper section (2. above) of the river Tisza twice per week (since 1995) between mid-April and mid-August. During our survey, we identified all Sand Martin holes and colonies and all Kingfisher holes. Because each species forms its entrance holes differently, the constructor's identity is straightforward. However, in large and dense Sand Martin colonies, it is difficult to pick out a Kingfisher hole from the mass of Sand Martin holes, but over a 15-year period, we have found that the Kingfisher is very strongly disinclined to breed in large and active Sand Martin colonies.

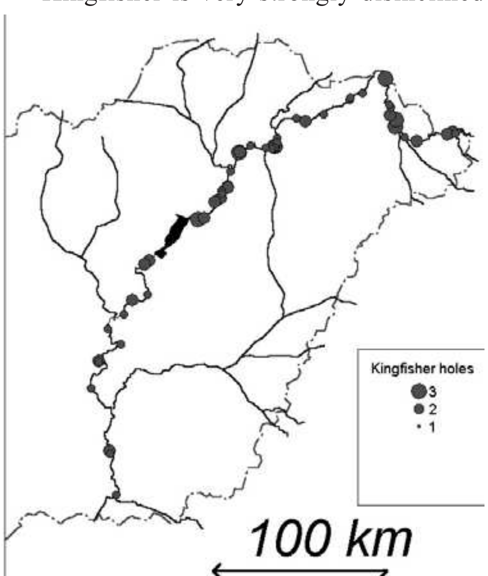

Fig. 8. Distribution of Kingfisher holes along the river Tisza in 2000. The circles indicate the location of the holes, and the circle diameters indicate the number of holes in one location (within 100 m).

The regular and intense spring floods (and more irregularly at other times) destroy most of the river walls where Sand Martin and Kingfisher breed. In most cases, previously used holes disappear or change in appearance, and so are easy to discriminate from new or reconstructed holes, making the annual surveys of active holes straightforward. Sand Martin population and colony sizes were estimated from the number of counted holes, the assumption being that 60% of the holes contain a nest (Szép 1990).

Such a simple technique does not work in estimating Kingfisher pairs along the river, because the male Kingfisher makes or re-uses many holes, most of which are not used during the breeding season. We carried out intense surveys along the upper section of the river between Tiszatelek and Tokaj (1997) and between Tuzsér and Tokaj (2000) when we checked all (100%) the potential breeding walls for Kingfisher holes, each hole being checked by endoscope for usage. In our analysis we handled separately the surveys of Kingfisher holes carried out, firstly of those of the Sand Martin colonies (1994-2000) and secondly that focussed on Kingfisher holes only (1997 & 2000). During the Sand Martin survey, the speed of the survey along the river is higher than in the case of the Kingfisher survey, which in the former could cause a lower detection rate for Kingfisher holes, for these are often in concealed places. For analysing trends in the Kingfisher population we consider the data collected during the Sand Martin survey, but only the detailed Kingfisher survey data were used for calibrating the frequency of nests in the holes. Holes with nests were identified when we found in them eggs or nestlings, or the remains of nestlings, faeces and fish scales. By this intense study we were able to identify in which holes Kingfishers were nesting in given years and we were able to estimate the ratio of holes with nest to excavated or reconstructed holes without nests. We present here the number of nests, because the estimation of breeding pairs from the number of holes with eggs and nestlings is confounded by the Kingfisher's opportunistic propensity for polygamy. Cramp (1990) indicates that one male may pair and breed with two, or more rarely three females. In 2000, after the ice had broken up, we mounted several surveys from motorboats in February and March along the upper section of the Tisza, to determine the presence of Kingfisher or to find evidence of fatalities from the cyanide pollution.

### 2.2. Surveying breeding success of the Sand Martin

Between 1995 and 2000, we checked twice a week 5617 Sand Martin holes with an endoscope. These holes were on randomly selected 2-5 m wide sections of 51 colonies along the upper Tisza or in sandpits adjacent to the river. This way we

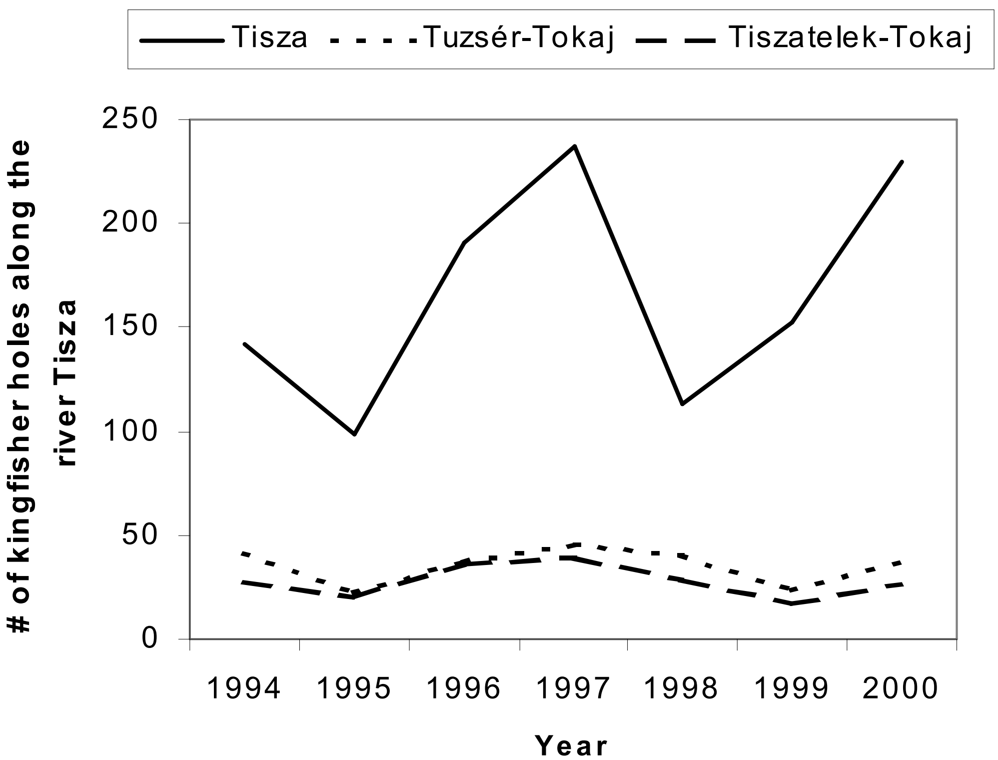


Fig 9. Number of Kingfisher holes along the river Tisza between 1994-2000. Three data sets are presented: a. Tisza: for the entire Hungarian section between Tiszabecs-Szeged (574 km) counted during the survey of Sand Martin. b. Tuzsér-Tokaj: along the Upper Tisza (72 km) counted during the detailed Kingfisher survey. c. Tiszatelek-Tokaj: along the Upper Tisza (42 km) counted during the detailed Kingfisher survey.

were able to identify the percentage of holes containing a nest (at least one egg laid), the start of the egg laying period, the clutch size, hatching success and the number of nestlings before fledging, which occured 14-18 days after hatching. The date of the first egg laid was recorded as being the number of days elapsed since 31 March (in the given year), day 1 being 01 April. Only those nests whose observations (of the above parameters) had been recorded with certainty had their data analysed. During the nest checking data analysis, we calculated and used the average values of the studied parameters for each colony, thus avoiding the statistical problems that arise from pseudoreplication.

### 2.3 Analysis of heavy metals in the feathers

We had collected one pair of the second outermost tail feathers from nine juvenile Sand Martins at a colony at Tiszatelek (48°12'N, 21°47'E) in June 1996, and we took the opportunity to do so again, from ten juveniles, in June 2000 after the pollution incident. The primary wing feathers of the sampled juveniles were undergoing moult, and so we could expect that these individuals had hatched and grown their tail feathers at the studied colony. The removal of one pair of tail feathers does not affect the bird's behaviour, survival nor dispersal characteristics measurably, and had been carried out under license of the appropriate nature conservation authorities. The removed feathers were stored in separate pairs in small sealed plastic bags, avoiding any feather contamination during collection and transportation, and were kept in boxes until the analysis. We used one tail feather of each pair for chemical analysis, and this was analysed but once, due to the small sample size. Concentrations of these elements were measured in the samples: As, Cd, Mg, Mn, Mo, Se, Sr, Co, Fe, Zn, Li, P, Ti, V, Ag, Cr, Ba, Hg, Pb, S, Ni and Cu. During this analysis, concentrations (in ppm) of control samples were examined, and these underwent the same handling protocols as did the samples (Vallner *et al.* 1999). The detailed descriptions of the preparation of the samples and the measurement of element concentrations (by inductively coupled plasma optical emission spectrometer, or ICP-OES) are given in Vallner *et al*. (1999, 2000).

### 2.4. Flood pattern along the river during 1995-2000

Along the Tisza the main flooding period is March-April, depending on the weather. The spring flood creates numerous extensive large new nesting walls. Flooding during the breeding period (called 'green flood') is rare and during the study years occurred only in 1998 when an exceptionally high level of flooding occurred between the end of June and mid-July. It covered and in most cases destroyed nearly all Sand Martin and Kingfisher nests along the river. Because this 'green' flood occurred before the main fledging period for both species, most first clutches were lost. The majority of the nest holes surveyed subsequent to the flood were holes that had been excavated after the flood, for second or replacement broods. In 1999, there was a moderate but long-lasting flood between April and June, which until mid-June had overtopped potential nesting places in the middle and lower Tisza.

After the pollution incident in 2000, we also needed to take into account the effects of the large spring flood, which submerged the potential breeding walls until early May along the Upper Tisza and until late May downstream. Early migrant Sand Martins were thus prevented from excavating nest holes and had to delay breeding. Statistical analysis of the data was carried out using the SPSS 9.0 statistical package (Norusis 1988).

## 3. Results

### 3.1 Population size and distribution of the Sand Martin

The population size changed markedly during 1990-2000 (Fig. 1). The average size of the breeding population along the Hungarian Tisza is 23 700 breeding pairs (bp) (SD=7761, n=10). In 2000, the population size increased by 102% in comparison to 1999, from 10 528 bp to 21 365 bp, which value was close to the average (Fig. 1).

We did not find a similar large increase along the upper Tisza, where no large changes have occurred since 1998. In contrast, this section in 2000 hosted a population whose size was only half (4746 bp) the average for the 1986-2000 period (mean=8985 bp, SD=3320, n=15) (Fig. 1).

The most significant factor influencing the large increase in population size along the Hungarian Tisza is the improved size and quality of prime nesting places in the form of perpendicular walls that had been created by the floods, mainly in the middle and lower sections (Fig. 2). In 2000, the total (vertical) surface of these riverbanks increased by 28%, from 4533 $m^2$ to 5820 $m^2$. Several large and substantial riverbanks had formed in the middle and lower section of the river (Fig. 2). It is important to bear in mind the adverse 1999 conditions that had prevailed, when the sustained flood along the middle and lower river sections had affected breeding to the extent that many pairs had been forced to breed in colonies remote from the river that year (Fig. 2). It is highly probable that many of these temporarily displaced breeding birds returned to the Tisza in 2000, thus explaining the large 'blip' in the annual breeding statistics.

### 3.2. Breeding success in Sand Martin colonies

An average of 63.82% of holes contained nest in the studied colonies (SD=16.20; n=51). This parameter does not show significant changes year on year (F=0.844; df= 5, 45; P=0.526; ANOVA) and it does not differ between Tisza colonies (mean=64.42; SD=15.95; n=32) and those remote from the river (mean=62.80; SD=17.01; n=19) (F=0.117; df=1, 49; P=0.734; ANOVA). The average value of the commencement of egg-laying is at day 53.08 (SD=6.39, n=51). This parameter does show significant changes year on year (Fig. 3) (F=3.146; df=5, 45; P=0.016; ANOVA) and it differs between Tisza colonies (mean=54.83; SD=4.93; n=32) and those remote from the river (mean=50.14; SD=7.54; n=19) as well (F=7.203; df=1, 49; P=0.010; ANOVA). Breeding began earlier in 1999 and 2000 than it did in earlier years (Fig. 3). In 2000, breeding in sandpits began earlier than in the river colonies (Fig. 3).

The average clutch size is 4.73 (SD=0.405; n=51). This parameter does not show significant changes year on year (Fig. 4) (F=1.718; df=5, 45; P=0.150; ANOVA) and neither does it differ significantly between Tisza colonies (mean=4.67; SD=0.429; n=32) and those remote from the river (mean=4.844; SD=0.346; n=19) (F=2.151; df=1, 49; P=0.149; ANOVA). However, in 2000 there was a slight difference; in colonies along the Tisza, the clutch size was smaller than at sandpit colonies (Fig. 4). The significant negative correlation between the start of egg-laying and clutch size at the studied colonies (r=-0.38; n=51; P=0.006; Pearson) could explain the lower clutch size at the delayed Tisza colonies.

The average hatching rate (at least one egg hatching per nest) is 0.749 (SD=0.24; n=51). This parameter does not show significant changes year on year (F=1.410; df=5, 45; P=0.239; ANOVA), nor does it differ between Tisza colonies (mean=0.721; SD=0.247; n=32) and those remote from the river (mean=0.796; SD=0.261; n=19) (F=1.071; df=1, 49; P=0.306; ANOVA).

The average value of the number prefledging nestlings is 3.265 (SD=1.329; n=42). This parameter does show significant changes year on year (Fig. 5) (F=7.363; df=5, 36; P<0.001; ANOVA). In general there is no significant difference between Tisza colonies (mean=2.970; SD=1.438; n=25) and those remote from the river (mean=3.70; SD=1.041; n=17) (F=3.219; df=1, 40; P=0.080; ANOVA). However in 1998, because of the 'green' flood, far fewer nestlings fledged from the Tisza colonies than from sandpit colonies (Fig. 5).

### 3.3. Accumulation of heavy metals in feathers of juvenile Sand Martins

Among the studied 23 chemical elements, only for magnesium (Fig. 6) did we find a significant increase from 1996 to 2000 for juvenile tail feathers collected at any one colony (1996: Mg=130.903 ppm; SD=39.474; n=9. 2000: Mg=196.976 ppm; SD=24.026; n=10; t=-4.461; df=17; P<0.001, t-test), but for lithium and nickel there were only weak increases (P=0.07). The amounts of titanium (Fig. 7) (1996: Ti=19.806 ppm; SD=14.59; n=9. 2000: Ti=7.570 ppm; SD=3.773; n=10; U=8; P=0.001, Mann-Whitney), cadmium (1996: Cd=0.767 ppm; SD=1.077; n=9. 2000: Cd=0.174; SD=0.09; n=10; U=17; P=0.022, Mann-Whitney) and selenium (1996: Se=10.518 ppm; SD=3.154; n=9. 2000: Se=6.991 ppm; SD=3.582; n=10; t=2.267; df=17; P=0.037, t-test) were significantly higher in the feathers collected in 1996 than in the 2000 feathers. For all other elements there were no significant differences between the two years of comparison.

### 3.4. Distribution of Kingfisher holes along the river Tisza

Kingfisher distribution is uneven along the Tisza (Fig. 8). The highest density of nest holes occurs on river sections that have the greatest incidence of breeding walls (mostly in the upper Tisza). In 2000, that the middle section held more nest holes than usual was very probably due to the severe 2000 floods creating additional large vertical banks suitable for nest holes.

### 3.5. Number of Kingfisher holes along the river Tisza

The number of Kingfisher holes along the studied 574 km-long Hungarian section of the river Tisza varied between 98 and 237 (mean=162.8 holes) (Fig. 9). The average density of the holes along the river is 0.29 holes/km (min: 0.17 holes/km; max: 0.41 holes/km).

### 3.6. Number of Kingfisher holes along the Upper Tisza

The number of Kingfisher holes along the 42km-long section of the Upper Tisza between Tiszatelek and Tokaj, where the intensive survey was done in 1997, varied between 17 and 39 (mean: 28.0 holes) (Fig. 9). The average density of the holes along this section is 0.67 holes/km (min: 0.41 holes/km; max: 0.93 holes/km). The number of Kingfisher holes along the 72 km-long section of the Upper Tisza between Tuzsér and Tokaj, where the intensive survey was done in 2000, varied between 22 and 46 (mean=35.1 holes) (Fig. 9). The average density of the holes along this section is 0.49 holes/km (min: 0.31 holes/km; max: 0.64 holes/km)

### 3.7. Frequency of holes containing nests

In 1997, out of the 74 Kingfisher holes investigated along the 42 km-long section of the Upper Tisza, breeding took place in 39, a nesting frequency of 52.7%. That year, nest density along this stretch was 0.93 nests/km. In 2000, out of the 37 holes investigated along the 72 km-long section of the Upper Tisza, breeding took place in 18 holes, a nesting frequency of 48.6%. That year nest density along the same (1997) section was only 0.29 nests/km.

### 3.8. Estimation of the Kingfisher population size along the river Tisza

Based on the detailed studies of 1997 and 2000, we can assume that *c*50% of the surveyed holes contain nests. From this value, we can estimate the number of nests varies between 119 and 49 (mean: 81 nests). The average nest density along the river is 0.15 nests/km (min: 0.09 nests/km; max: 0.2 nests/km). Along the Upper Tisza, nest density could reach 0.5 nests/km.

### 3.9. Kingfisher breeding season in 2000

We did not find any evidence of pollution-induced fatalities during our visits before and during the breeding season. We found that there were considerable delays in the arrival of prospecting birds and in the start of breeding, but the lengthy and heavy flood in 2000 was very probably the cause.

## 4. Discussion

The sudden and heavy cyanide pollution along the river Tisza in February 2000 caused mass fatalities, amounting to several tonnes of fish and created serious concern about the potential impact, from the immediate to the long-term, on the unique, highly natural ecosystem of this river. Because this pollution occurred at the biologically most inactive period of the year in this area, the overall damage, though considerable to local invertebrate and fish populations, was probably much less than it would have been at any other time of year. However, given precedents such as the similar disaster at Donana, it is reasonable to expect further and serious delayed impacts on the fauna and flora of this highly natural riverside ecosystem, whose diversity and abundance in the temperate European biome is almost unrivalled in scope. Had the cyanide pollution occurred only one month later, heavy metal pollution could have had the most devastating effect on the migrant and resident breeding fauna whose food supply is almost totally dependent on the immense diversity and abundance of invertebrate (mainly insects) and vertebrate (mainly fish) species. Only by the narrowest of margins (one month) did severe heavy metal poisoning fail to spread throughout this precious habitat in the spring-induced annual acceleration of biological activity. A vigorous spring flood did much to replenish the losses. The diverse and abundant bird fauna along the river is potentially one of the most vulnerable groups under threat from pollution incidents such as this. The very vulnerability of this group, through its strong dependence on the quality of its breeding environment, makes it one of the best candidates to indicate the negative effects of pollution; in other words, it contains ideal species to act as monitors of environmental quality.

The long-term integrated monitoring of the insectivorous Sand Martin population along the river Tisza could provide a near-ideal source of analysing the negative delayed effects of pollution, not only because the food supply of the studied breeding population is strongly related to the diversity and abundance of the invertebrate fauna, but also because detailed long-term data are available for the analysis.

The intensive studies on this population did not reveal any negative effects of the pollution on population size, distribution, and breeding performance in 2000. We found only that the clutch size in that part of the population along the upper Tisza was smaller than in populations breeding far from the river, but this difference very probably was due to the 2000 flood. One important effect of that unseasonably lengthy flooding was that only the later returning migrants were able to breed along the river (the early migrants having sought other sites), as is shown by the breeding data start dates. Later returning migrants usually have a smaller clutch size (Moller 1994), which proved to be the case with the Tisza Sand Martin population in 2000, as shown by the negative correlation between the start of egg laying and clutch size.

From our chemical analysis of tail feathers of juvenile birds, we could not show that the pollution-deposited heavy metals caused changes in the level of those heavy metals in the food chain of insectivorous birds. However, comparisons of these feathers with feathers grown in 1996 suggest that in earlier decades levels of various heavy metals were higher in this group's food chain, thus correlating with information that the polluting industries in the Romania-Ukraine watershed region had been working at higher production rates than recently.

The Sand Martin population size increased significantly in 2000, as did the species' distribution along the river, but we need to probe beyond these findings to consider which other factors might have compensated for, or hidden the effects of the pollution.

Earlier studies of the effects on survival rate of conditions en route and in the wintering areas of this long distance migrant species revealed that survival rate and, in part, the population size are highly dependent on the weather they experienced in Africa (Szép 1995). The 1999-2000 migration and wintering season was more typical than average, because of generally favourable weather along the migratory route in Africa. The very early return to the breeding area in 2000, allowing breeding to begin early, is an indicator of good wintering conditions. Such favourable circumstances would improve not only the likelihood of an increase in the breeding population, but also that breeding individuals would be in good average physical condition.

Flooding along the Tisza has a number of positive and negative impacts on the breeding population. Flooding mainly occurs in March-April and has a very important positive effect by creating a fresh pristine perpendicular riverbanks for breeding colonies. That walls with previously occupied holes (that still contain several active ectoparasite species) are washed away regularly is a very important factor in reducing or eliminating ectoparasite load, thus minimizing its high cost and threat to individuals and colonies (Szép & Moller 1999, 2000). Heavy and sustained flooding, as in 2000, can create new and extensive breeding habitats in those stretches of the river that normally hold few or mainly small walls. Flooding occurring during the fish breeding period has an important positive role in the reproductive success of several fish species and possibly is beneficial for several other aquatic animals. The 2000 flood was very important because it diminished the effects in the polluted areas by the widespread translocation of numerous non-vertebrate and vertebrate species from the adjacent uncontaminated wetland habitats of the Tisza floodplain.

However, flooding during the birds' breeding period could have several adverse impacts. Flooding that extends beyond the normal spring period, as happened in 1999 and to a lesser extent in 2000 makes it impossible for early returning Sand Martins and Kingfishers to breed, and this causes a large scale dispersal to secondary breeding habitat in sandpits where natural threats (digging predators, wall collapse, lower food availability in windy and cold weather) and human impact (illegal sand removal) are greater than along the river. Flooding occurring when nestlings are feeding could wipe out most first clutches in the breeding population, which natural disaster happened in 1998. Historically, the frequency of such abnormal flooding has been low but recently, the incidence has been much higher, and if this change is sustained, the putative link to climate change becomes more substantive.

From these studies, we found that along the river Tisza a large population of Kingfisher breeds whose size varied between 50-120 bp, if polygamy remained at a low level. This population density approaches the highest values cited in the literature (Cramp 1990) reaching 0.5 bp/km in some sections.

Considering and comparing the population processes before the 2000 pollution and those just after, we could not find immediate post-pollution serious effects on Kingfisher. We did not find evidence of fatalities caused directly by pollution. To explain the low population size and density found in 2000, which were very low in comparison with 1997 data (the first detailed study), we can show that the primary causes were the effects of the flood of 1998 on reproduction and of the flood in 2000 on delaying the onset of breeding; the pollution incident had no such immediate effects. When we consider the huge importance to the Kingfisher of small fish as food, we can show that the pollution did not cause a large negative impact on this piscivorous species.

Based on our experience along the river and on the results since 1995 from our intensive fieldwork in the breeding seasons, we know that some cases of several *Chyronomous spp* species swarming in 2000 were particularly abnormal in their abundance and time schedule, in that they were not typical of the prevailing conditions, whereas they had been so in the previous decade. These odd circumstances, by providing a potentially ample food supply for insectivorous birds, could explain the very successful breeding performance at the colonies along the river, where although the clutch size was smaller than at sandpit colonies, there was no difference in the numbers of nestlings produced. There is no research evidence for any connection between the pollution and the abnormal swarming. However, neither can the role of pollution be excluded as contributory to the abnormal swarming, which also has had positive effects on insectivorous animals in 2001. It may be that this perturbation may have longer-term negative impacts.

In summary, our study, which has the longest-term data series for the entire Hungarian section of the Tisza, did not find evidence for any negative impact in 2000 of the pollution on breeding insectivorous and piscivorous bird species. However, that does not preclude a long-term, insidious perturbation of the Tisza's ecological system, as the abnormal *Chyronomous spp* swarming behaviour might indicate. This conclusion accords with the results of investigation of other animal groups, but much further work is to determine what, if any, the medium- and long-term middle and long-term effects of the pollution are.

Our work underlines the importance of detailed and long-term investigation of bird species that can act as indicators of environmental change, particularly that suffered through the impact of human-induced-made environmental disasters. Such impacts are likely to be quickly obvious where large groups of breeding insectivorous and piscivorous birds occupy the course of rivers possessing numerous extensive natural habitats of high biodiversity and biological productivity. Work such as ours can discriminate between natural and human-induced causes of bird population changes, and without this ability, it is all too easy to reach the wrong conclusions. It is important to reiterate that these types of studies provide one the most cost-effective, large scale and long-term opportunities to investigate the effects of pollution on insects and fish, the other two main groups of animals that inhabit the European riverine environment.

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**Ornis Hungarica 12-13: 183-190. 2003**

# Swallow censuses in Northwest Germany (1986, 1991 and 1996)

H. Oelke

Oelke, H. 2003. Swallow censuses in Northwest Germany (1986,1991 and 1996). – Ornis Hung. 12-13: 183-190.

Starting in 1961, from 50-100 villages and settlements in the industrial area of Hannover-Braunschweig (centred on Peine) have been checked at 5-year intervals to count the nesting population of House Martin *Delichon urbica* and Barn Swallow *Hirundo rustica*, the method being a direct nest count. The hitherto stable Barn Swallow populations declined in the late 1980s whereas the steady increase of House Martin ceased because of new developments following German reunification in 1989. The numerical results are presented. Decrease and increase are analysed in regard to landscape (hillside - moraine - loess zones), size and economic status of the settlements (between 50-50 000 inhabitants, traditional rural - highly industrialised areas).

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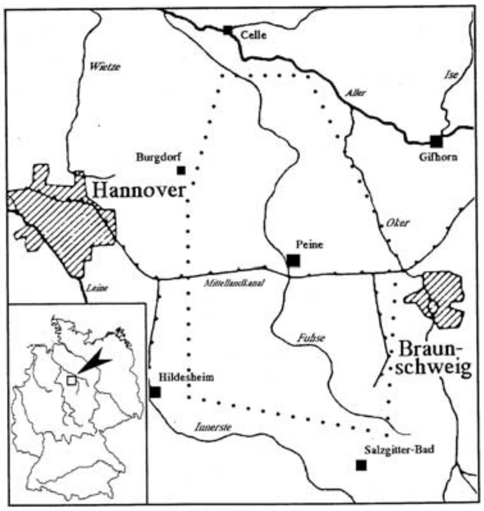
## 1. Introduction

For my thesis (Oelke 1961) on the abundance and ecology of bird populations within the central German Peiner moraine and loess area (Hannover-Brunswick) (Map 1), a stretch between plain and hillside and mountainous sectors, I had to solve the problem of how to describe bird densities for species whose territories could not be mapped and which occupied complex habitats not easily accessible. These circumstances applied particularly for swallows in widespread and complex human settlements (the city of Peine and numerous villages then expanding) whose population sizes ranged from only 10 up to 30 000 people (Figs 2, 3). My job as a teacher of biology and chemistry helped me to persuade students and classes to count occupied swallow nests (Method: total count). The same procedure was followed by a several amenable and cooperative colleagues from other schools, mostly primary schools (grades 3-4), some secondary schools (grades 5-10) and a few high schools (German 'gymnasia') (grades 11-13) in the approximate 800 km$^2$ of the study area. In the early years of the project, elderly and retired teachers (known as Dorfschullehrer) who were still active (born around 1880-1920) assisted, using their organizing and instructing experience, and their devotion to helping out in their home village. Even at the beginning of the swallow counts, members of the Peine Nature club (Peiner Biologische Arbeitsgemeinschaft of 1953) were cooperating and supporting the project eagerly.

The success of the first count in1961 (Oelke 1962, 1963) brought about the decision to repeat the counts at 5-year intervals. These later counts were documented as follows: 1966 (2nd count), (Schierer 1968),




Map 1. The study area 'Peiner Moränen- und Lößgebiet' (dashed line) and its location in Germany. From Oelke & Heuer (1993).

1971 (3rd count), (Oelke & Tinius 1973), 1976 (4th count), (Oelke 1981) and 1981 (5th count), (Oelke & Schütze 1985). For reasons of stress, political involvement and health, the counts for 1986 (6th count), 1991 (7th count) and 1996 (8th count) have not yet been summarized and published, but are presented here. The preparations for the 9th count (2001) have already started.

The main conclusions of the 1961-1981 counts may be presented thus: Barn Swallows (*Hirundo rustica*) are concentrated in rural places with good cattle stock, where the sheds and storehouses are larger, and where there are traditional, conventional farms and farm buildings. House Martins (*Delichon urbica*) are concentrated in buildings of the post-war construction period built towards the open edge of the urban periphery where there is first reduced, and then younger tree and shrub cover. Common to both swallow species is that their breeding concentrations are fairly adjacent to wetlands, river marshes,

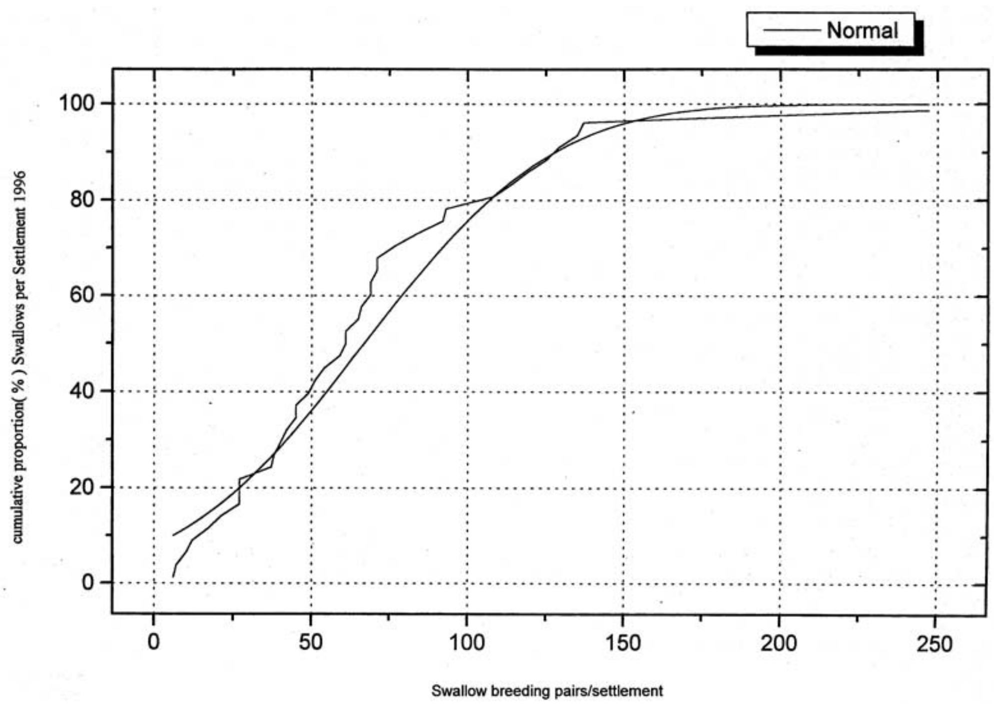



Fig. 2. Cumulative proportion (%) breeding swallows per human settlement to total breeding pairs per human settlement 1996.

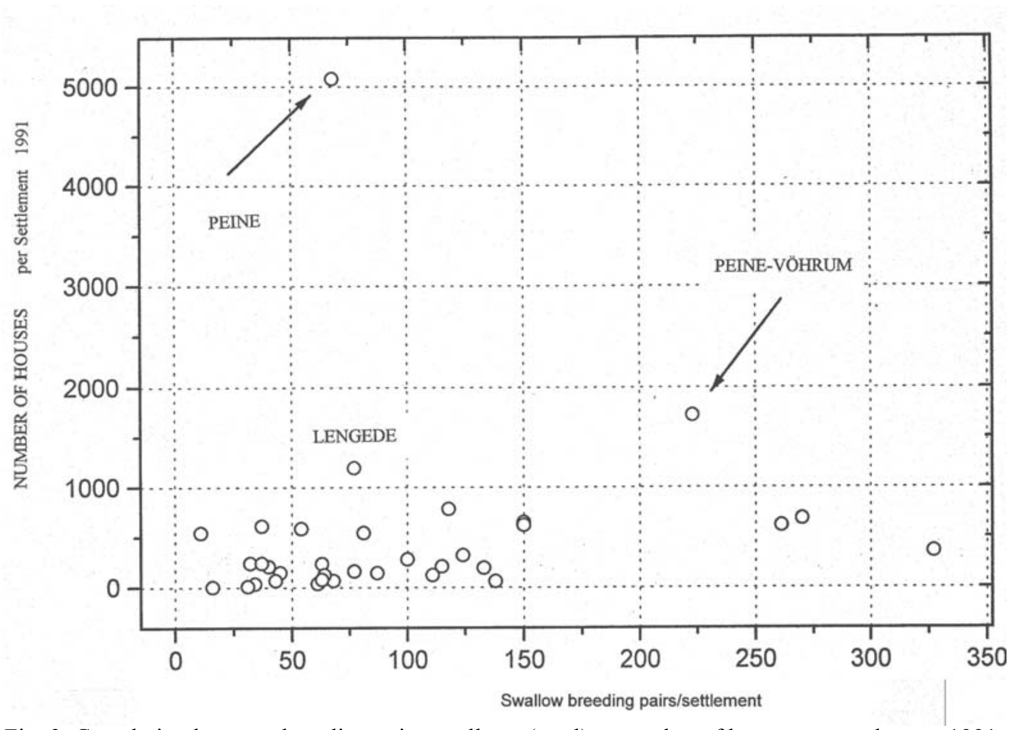



Fig. 3. Correlation between breeding pairs swallows (total) to number of houses per settlement 1991.

large open areas and stretches of pasture and sizeable extended woodlands, especially at where the typical hilly country of the Central German mountains begins. The wide varieties of buildings and of settlements produce very individual landscapes. Any form of industrialisation, such as the iron melting plants and steel factories of the Peine-Salzgitter industrial area, their conglomerates of satellite industries, the web of transportation systems, the zones of oil refineries and chemical plants (Dollbergen, Peine, Lehrte), and the wide agricultural plains (characterised by modern spray and fertilizer agriculture as in the loess zone within the Hildesheim-Peine-Salzgitter-Wolfenbüttel, or 'Börde', area), has depressed and nearly swept away swallow populations for much of the period before 1986. The cost of the agricultural revolution took the form of loss of nesting and feeding habitats, especially for *Hirundo rustica*.

The Barn Swallow decline became clear in 1981. House Martins had been increasing up to that time mainly because of the private house building boom of the 1970s and 1980s. However, up to 1961 no settlement had been completely deserted by the two swallow species.

The Peine swallow counts (to describe them simply) have been adopted in some places elsewhere in Germany and even in France, but as far as I know they have not been reproduced over such a timescale nor in such a large area.

## 2. Methods and material

All censuses have used the 1961 methods without deviation. The central school administration distributed a special instruction form early in the year to all school headmasters and to the membership of the Peine Nature Club and neighbourhood nature clubs (NABU, BUND). The project was publicly announced in local and regional newspapers. Participants in former counts were asked to continue. The information was distributed as early as possible in the census year to arrive before the summer vacation. Any interested person encountered by the organizers was asked directly to participate.

The number of participating counters varied from >151 (1986) to >182 (1981) and >73 (1991), a total of at least 400. The reduction in 1991 arose through many schools withdrawing: the teachers found themselves unable to cope with the increased workload that accompanied widespread school reform, and so many were unable to continue organizing their part in the swallow project. Furthermore, there was an amazing change in student behaviour, where peer pressure made them react to systematic and cooperative fieldwork with fear of ridicule. A considerable hiatus, not fully overcome, was caused by the death of 8 elderly and long-serving counters (see Acknowledgements). In many cases I tried to fill the gap myself and visited the more distant villages by car, thereon using a bicycle or walking from street to street and from house to house, wary of the dangers of aggressive dogs. Through experience, nest sites and colonies in older structures were sought out. In the more spread out streets and blocks, the bicycle came into its own.. For the welfare of the birds and to leave the nests undisturbed, there was no direct analysis of nests, ie there were no counts of eggs and clutches, nor was any ringing attempted. Data from nest analyses from previous studies are as yet unpublished.. It became apparent that a complete census account was not possible, nor were absolute counts achievable from census to census. Testing my students (aged 18-22) in the village of Oberg, (popn. 2200 in 1996) revealed that the hidden error rate was 20%. (For phenology, see Fig. 7, (the arrival of barn swallows in the tiny village of Warmse on the alluvial river Aller plain).

Statistical data and weather characteristics were contributed by the communities and by the office of statistics in Hannover, the capital of the federal state of Lower Saxony or otherwise were obtained from the literature. There was no extreme weather to disturb the 3 breeding seasons of 1986, 1991 and 1996. No public financial support was ever granted for this project. All expenses of the initiative were covered solely by private expenditure.

## 3. Results

1. All samples (settlements)

The main results are summarized using the following parameters:

| | 1986 | 1991 | 1996 |
|---|---|---|---|
| Surveyed settlements | 108 | 93 | 51 |
| Breeding pairs of *Hirundo rustica* | 3598 | 2985 | 1473 |
| 1986 = 100 | 100 | 83 | 41 |
| Breeding pairs of *Delichon urbica* | 4971 | 3666 | 1731 |
| 1986 = 100 | 100 | 74 | 35 |
| Total number of breeding swallows | 8569 | 6651 | 3204 |
| 1986 = 100 | 100 | 78 | 37 |

These figures suggest an alarming decrease of swallows in the study area. However, they have to be corrected to incorporate all settlements surveyed in the 3 study years. The number of counters reduced continuously from 1986 onwards making it impossible to survey simultaneously all the studied locations (ie over 123 since the beginning of the project).

2. Samples common to all 3 study years

Breeding areas surveyed in an identical manner in the years 1986-1996 (n = 44)

| | 1986 | 1991 | 1996 |
|---|---|---|---|
| Breeding pairs of *Hirundo rustica* | 1346 | 1403 | 1279 |
| 1986 = 100 | 100 | 104 | 95 |
| Breeding pairs of *Delichon urbica* | 2254 | 1996 | 1683 |
| 1986 = 100 | 100 | 88 | 75 |

These results indicate that the regional swallow populations are stable (with no significant decrease or increase in Barn Swallow numbers). However, they point to a slight decrease in House Martins between 1986 and 1996 (*P*<0.05).

There are remarkable differences in the nest counts of different breeding areas and counting periods:

| Changes[2] | 1986-1991 | 1991-1996 | 1986-1996 |
|---|---|---|---|
| Hirundo rustica | | | |
| Decrease (>20 %) | 28 | 11 | 25 |
| Increase (>20 %) | 15 | 1 | 11 |
| Stability (± 20 %) | 6 | 1 | 7 |
| Delichon urbica | | | |
| Decrease (>20 %) | 22 | 10 | 12 |
| Increase (>20 %) | 20 | 2 | 3 |
| Stability (±20 %) | 6 | 1 | 1 |

[2] Number of settlements with different forms of variations

Characteristics of the 44 comparable breeding areas

| | 1986 | 1991 | 1996 |
|---|---|---|---|
| Maximum number of breeding pairs (bp)/settlement | 250 | 270 | 247 |
| Minimum bp/study area (village) | 6 | 9 | 6 |
| Mean | 83.93 | 79.07 | 65.7 |
| Median | 72.0 | 63.5 | 58.0 |
| Variance | 3214.25 | 3700.20 | 2381.74 |
| Standard deviation | 56.694365 | 60.829307 | 48.803154 |
| Human population | | >80 000 | |
| Number of houses | | >17 000 | |

The changes seem to be balanced. Highly significant statistical differences (*P*<0.001) could be proved (WINSTAT statistical program for Windows, Springer 1998) for the Barn Swallow (for 1991/1996) and for the House Martin (for 1986/1996 and 1991/1996). The House Martin has suffered most in recent years, possibly because of the slow-down and

Fig. 7. Spring arrival (first observation) of Barn Swallows in the farm complex of Warmse, Samtgemeinde Meinersen, county of Gifhorn 1969-2000. Copyright Hans Oelke, March 15, 2001.

even cessation of the building boom and the change from multi-storey blocks containing rented flats to smaller villa-like privately owned houses. There have been many cases of house owners and residents removing House Martin nests or discouraging nesting in all sorts of ways, such as by putting up in front of suitable nesting sites strings, narrow boards, nets, toys and plastic shields, or by installing dark-coloured coving and sealing off overhanging roofs. This way, they hope to avoid any staining and soiling on their windows, house walls and balconies. Barn Swallow nest sites were affected mostly by repair, painting, spraying and cleaning sheds in spring, thus removing or sealing off old nest sites. Open and freely accessible sheds are becoming rarer and rarer. The swallows' former habit of breeding in isolated cattle sheds in pastures or under small, wooden river bridges (as on the rivers Fuhse and Erse) has completely ceased. In the future, concern about the possible transmission of highly infectious livestock diseases such as BSE and foot and mouth disease might require stables and barns to be sealed off from entry by swallows.

3. Correlation: size of human settlements

(population size) and swallow populations (total bp of *Hirundo rustica* and *Delichon urbica*)

The majority of local swallow populations occur in human settlements with 10-5000 inhabitants (Map 1). Larger villages and the provincial town of Peine have disproportionately low swallow occupation (Figs 2, 3 ). The same holds true when considering the whole range of swallow population sizes (Fig. 2). 250 House Martin and Barn Swallow bp are the maximum for these villages and for urban settlements. There is an inverse relationship between the number of houses per human settlement and the number of breeding swallows (Fig. 3).

Both parameters (houses and swallows) correspond in a straight and algebraic regression (r=04154, $P$<0.001) when urban areas are omitted (calculated for 1991).

4. Special census examples from nearly complete census series (1961-1996)

Fig. 6 illustrates different forms of Barn Swallow and House Martin long-term population changes. In my home town of Peine, (population size: *c*26 000 and in Salzgitter-Lichtenberg (Fig. 5) (a satellite suburb (population size: 3000) of Salzgitter city), both the dominant House Martins and the subdominant Barn Swallows are decreasing significantly. Two neighbouring villages Gross Ilsede (population size: 2600) and Oberg (population size: 2200) differ markedly in the density and development of the two swallow species. House Martins have increased in Oberg and decreased in Gross Ilsede. Ohlum (population size: 235 pairs) (Fig. 5)

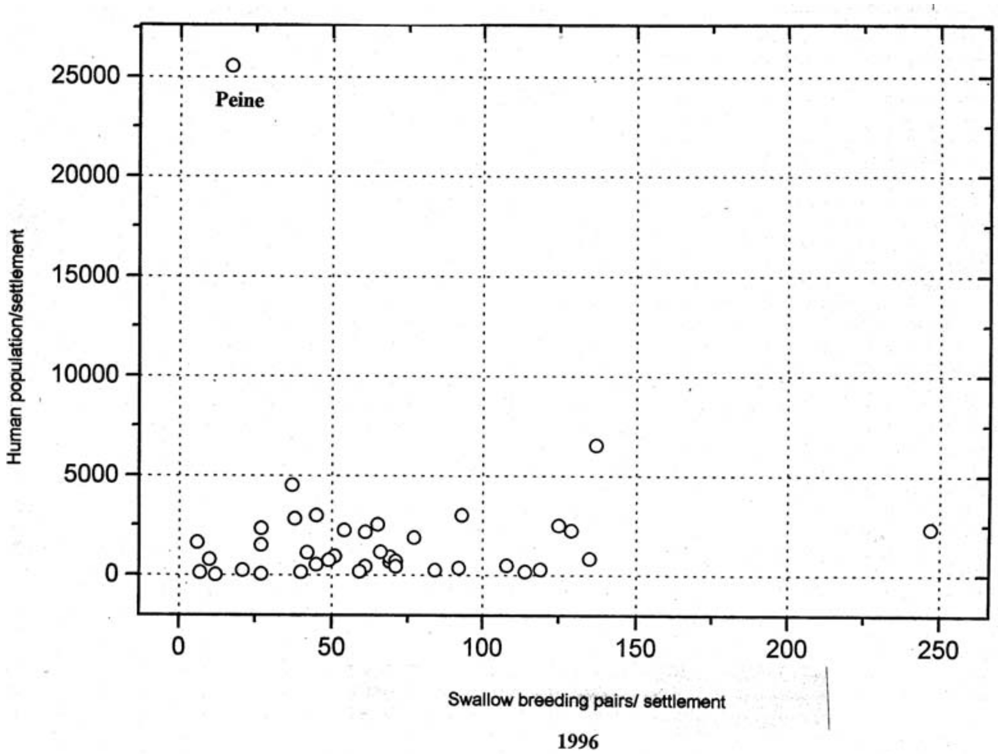



Fig. 1. Correlation between human population size and swallow breeding populations (total swallow breeding pairs per human settlement).

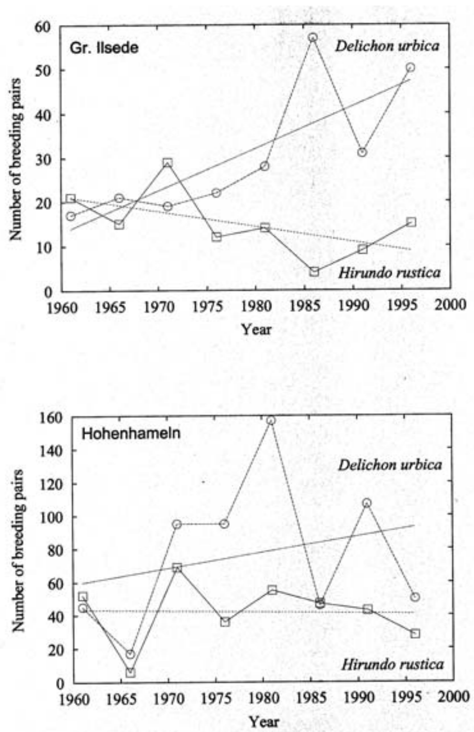



Fig. 4. Development of Barn Swallows and House Martins in Gross Ilsede and Hohenhameln 1961-1996.

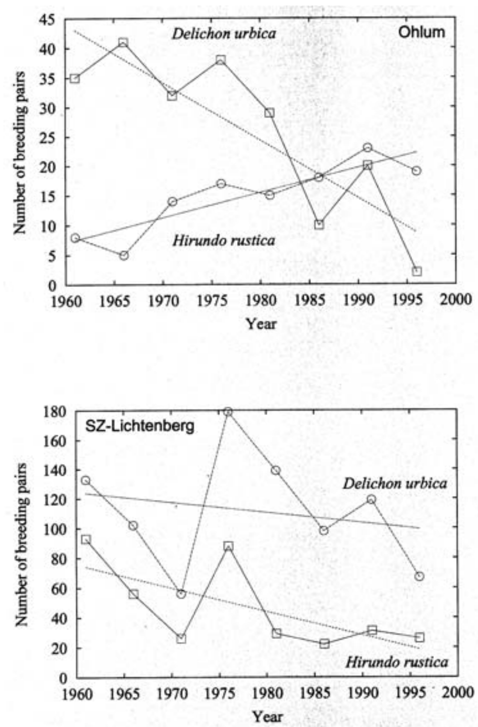



Fig. 5. Development of Barn Swallows and House Martins in Ohlum and Salzgitter-Lichtenberg 1961-1996.

in the loess zone has lost its Barn Swallows while Wiedenrode (population size: 176) in the glacial river plain of the river Aller and once the village that had the proportionately highest count of House Martins, has witnessed their decline. The villages of Hohenhameln (population size: 2900) (Fig. 4) in the loess zone, and Edemissen (population size: 2550) in the moraine zone might have been the first to note the swallows' decline, although both swallow species withstood extinction for a long period. A balanced swallow population characterizes the village of Abbensen (population size: 1740). The fluctuations that occurred at the tiny farm complex of Warmse (population size <60) (Fig. 6) in the glacial Aller plain are remarkable, for although this hamlet has now lost nearly all its swallows, one family (Mr. Gottschalk, Mrs. Halberstadt) still keep a record of the arrival of Barn Swallows. Their recording began in 1969 and has continued up to 2000. (Figs 1, 2), revealing the mean arrival to be April 18±2 days. 95 % of all swallows return between April 14 and April 22.

## 4. Conclusions

1. The long-term nest counts of swallows in northwest Germany should be interpreted - as models of the situation and the trends over the larger regional bird populations treated as a meta-population.
2. Surveys of smaller fractions of this population are not representative of the

situation of swallow species in larger geographical areas.

3. Despite all the decreases, Barn Swallows and House Martins heading for extinction even in such a highly industrialised area as Lower Saxony in Germany.
4. The methodological problems of covering and analysing a large proportion of a widely distributed bird species are immense and are costly in time, labour input and expenditure. The Peine swallow project might be able to continue using but a few suitably selected locations from all the human settlements, studied over the last 40 years.
5. Schoolchildren and younger students must be superseded by adults with more application and who enjoy systematic studies.
6. In future, public support will be more difficult to achieve, as will active assistance from farmers.
7. Only a fraction of the available registrations have been returned, leaving many evaluations to be undertaken.

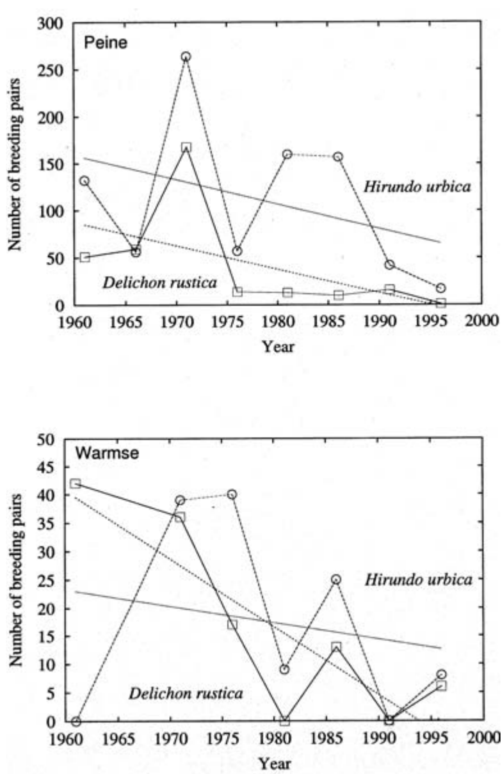



Fig. 6. Development of Barn Swallows and House Martins in Peine and Warmse 1961-1996.

*Acknowledgements*. I cordially thank the census participants of 1986, 1991 and 1996, a complete list of whom will be published separately in a German version of this paper. Special thanks must be paid to Dr Ludwig Schweitzer (for statistical help), Kai Stich (for programming) and Peter Becker (for replacing lost literature). My former secretary Elly Müller from Peine took good care of the huge correspondence, especially during the last 3 censuses. In memoriam of Herbert Fette (†1992), Sascha Gutneder (†1992), Arthur Heinken (†2001), Georg Köstermann (†1994 ), Oskar Kroß (†1997), Otto Meier (†1995), Karl-Friedrich Schmidt (†1996), Hermann Schwenke (†1995) who contributed decisively to these results.

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**Ornis Hungarica 12-13: 191-196. 2003**

# Monitoring Aquatic Warbler *Acrocephalus paludicola* in Poland

J. Krogulec and J. Kloskowski

Krogulec, J. and Kloskowski, J. 2003. Monitoring Aquatic Warbler *Acrocephalus paludicola* in Poland. – Ornis Hung. 12-13: 191-196.

As a globally threatened species, the Aquatic Warbler requires a monitoring programme for population trends to be tracked. Such a programme should consider the timing of censuses to achieve precise assessments of the number of the singing males within the limited singing period. We compare results of whole-population counts conducted in the southern Biebrza basin between 1995 and 1997 and in the Lublin area between 1993 and 1997. Densities of singing males showed different trends for selected fen mire patches within the same wetland complexes. Based on the results of the nationwide census in Poland (1997), we compare data collected during two breeding season counts (between 20 May-6 June and 17 June-3 July respectively) relevant to the timing of the first and second breeding attempts. On most census plots we recorded significantly higher numbers of singing males during the second count than in the first count. Increases in the numbers of singing males were recorded mainly in areas burnt or flooded in spring. In earlier surveys, such places often had been considered 'inappropriate' for the Aquatic Warbler. We discuss some hypotheses concerning changes in the numbers of singing males between the first and second brood surveys. Extending the time of the second count until late June may be important in planning future conservation actions. Also, habitat features during second breeding attempts should be taken into account in the management of the breeding areas.

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## 1. Introduction

The Aquatic Warbler is a bird species threatened on a global scale (Tucker & Heath 1994), and Poland holds a considerable proportion of its population (Aquatic Warbler Conservation Team 1999). Consequently, protection of this species in Poland is of prime importance for its existence as a whole. The first estimate of the population status, distribution and abundance of the Aquatic Warbler in selected territories it occupies in Poland was prepared by Dyrcz & Czeraszkiewicz (1993). However, this estimate was based on extrapolations of counts from transects in larger areas and it omitted the major Aquatic Warbler breeding site, the Biebrza marshes. Complete censuses of whole populations were performed in the Lublin area (calcareous Chelm marshes and Poleski National Park) in 1993 and in the southern Biebrza basin in 1995. In 1997, the first single-year countrywide survey was undertaken, except for the southern Biebrza basin, where only some selected sub-areas were counted (see Aquatic Warbler Conservation Team [1999] for details). During the latter census covering all the important breeding areas, two counts were conducted at a minimum

Tab. 1. Population estimates and major habitat type within the main Aquatic Warbler breeding areas in Poland.

| Site | Population estimate | Major habitat type |
|---|---|---|
| Western Pomerania | 230 | Seasonally-flooded brackish marshes, reedbeds |
| Biebrza and Narew marshes | 2040-2080 | Rich floodplain marshes in river valleys, open sedge fen mires and wet marshy grasslands covered by high grass and clumps of sedge |
| Lublin Province marshes | 450 | Calcareous marshes with *Cladium mariscus* |
| Other sites | 180 – 190 | Small remnants of marshes overgrown by reeds, reedbeds |
| Total | 2900 – 2950 | |

interval of 2 weeks, to parallel the timing of the first and second broods. In this paper we compare the results of the counts between years and between the first and second brood counts.

## 2. Study area and methods

In all years, the censuses were carried out during the peak period of male activity between 15 May and 30 June. All complete censuses (those covering all apparently suitable habitats in each geographical area) were performed in the same way. The surveyed sites were divided into sub-areas of c100-300ha, which could be counted within 2 hours around sunset by a group of 4-5 observers. Natural boundaries, like ditches or forest patches, demarcated the sub-areas. During the counts, the participants were arranged in an extended line within sight of one another (50-100m separation). Each observer recorded the singing males found on the stripe between himself and one of his neighbours. For each censused sub-area, data were gathered on land use (mowing, grazing) and recent floods and grass burnings (for more details see Kloskowski & Krogulec 1999).

Most sub-areas were surveyed twice at intervals of at least 2 weeks during the 1997 countrywide census in western Pomerania, the southern and central Biebrza basin and in the Lublin area (calcareous marshes near Chełm and fen mires in the Poleski National Park). Subsequently, for each geographical area we compared the results from the first and second visits. Only counts conducted between 20 May and 6 June for the first broods and 17 June and 3 July for second broods were included in the analyses, except for some from areas in Western Pomerania, where the counts were relatively late, the first having been delayed until 13 June.

## 3. Results

Overall in 1997, in the first count made which we consider to be the basic count for estimating the number of singing male Aquatic Warbler in Poland, 1307-1341 singing males were counted. During the survey of the Aquatic Warbler in 1995 in the Biebrza southern Basin, the 1578-1609 males were detected. Upon adding these two numbers, the population of the entire country may be estimated at 2885-2950 males. Acceptable round figures for Poland are 2900-3000 singing males. Detailed data on population estimate and major habitat types within the main Aquatic Warbler breeding areas in Poland are given in Tab. 1.

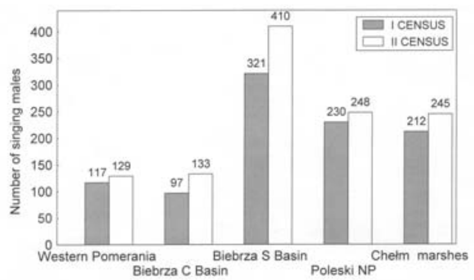


Fig 1. Numbers of singing males during the first and second censuses in the most important Aquatic Warbler breeding areas in Poland, 1997.

**Comparison of first and second censuses**

In all the important breeding areas of the Aquatic Warbler in Poland where double counts were conducted systematically (in over 50% of the monitored areas), the numbers of singing males increased between the first and second census (Wilcoxon signed-ranks test, $T_{10}$=0, n=5, *P*=0.043; Fig. 1). However, trends in numbers of singing males varied between the sub-areas delineated for census purposes within the same complexes of marshlands.

Although our data on habitat quality were not sufficient for quantitative assessment, in some areas the increase in the number of singing males during the second censuses was clearly associated with habitat improvement in the course of the breeding season. These habitat improvements usually comprised areas burned or flooded in the spring of that year where the vegetation had not regenerated before the second broods had been started. Elsewhere, in a few plots adjacent to human settlements, grazing by domestic cattle could have thinned out the dense layer of lower vegetation.

## 4. Discussion

### 4.1. Differences between first and second censuses

On the basis of the previous research (Dyrcz & Zdunek 1993) it has been assumed that the density of breeding males is at its highest during the first breeding attempts (the second half of May and the first week of June) and then gradually decreases. However, during the 1997 countrywide census, numbers of singing males were higher during the second counts than during the first counts in all the important breeding areas. This pattern could have at least four different explanations, which are not exclusive:

1. The differences result from exchanges between nearby subpopulations (Schaefer *et al*. 2000) and the overall increase in numbers is an artifact of these changes in density in particular areas. Similar high numbers were observed during the 1998 July counts in some plots in Sporova and in the Zvanets mires in Belarus. Here the reasons for the apparent movements of the birds between the plots were spring burning of vegetation and summer floods caused by abundant precipitation (Kozulin & Flade 1999). However, the overall increase in numbers in Poland was consistent in all studied areas and we believe that it is unlikely that any important breeding areas of Aquatic Warbler that might have been a source of the males' influx have been left undetected in Poland.
2. Breeding Aquatic Warblers settle initially in a more dispersed manner, occupying marginal habitats dispersed

around the 'core' areas. Some marginal habitats might have been occupied unrecorded, having been assessed as 'unsuitable'. Later in the season, when the birds are more aware of breeding conditions (and there is availability of receptive females), the males cluster in the most optimal areas. Small numbers of singing males were recorded in some isolated areas that may previously have constituted important breeding habitats, but have suffered a significant degradation, *e.g.* the Ner river mouth (Aquatic Warbler Conservation Team 1999).

3. The number of singing males during second broods in marshlands of Poland increases because males from breeding areas east of the Polish border (in Belarus and Ukraine) leave these grounds and gather in Poland, where they attempt to mate.
4. The timing of the spring arrival at the breeding grounds is widely spread between males (see also Persson & Öhrström 1989), some of them arriving too late to compete for females undertaking the first brood. Alternatively (in the case of young males) intensified advertising for females occurs later in the season because they need time to attain post-migration recovery. This may be important, because females seem to return from their wintering quarters not much later than males (Dyrcz & Zdunek 1993).

It must be noted that for logistical reasons the dates of our counts were spread over more than two weeks each and did not match perfectly the peaks of laying the first eggs of the first and second clutches, 15-29 May and 21 June-5 July respectively (Dyrcz & Zdunek 1993). However, of particular note is that the initiation of second clutches appears to be poorly synchronized and may differ between breeding areas within the country and between years.

The ultimate verification of the above hypotheses will be difficult, despite thorough ecological and behavioural research in recent years. Some basic information on the breeding biology of Aquatic Warbler, such as breeding sex ratio, site fidelity, inter- and intra-sexual differences in the timing of arrival on the breeding grounds, is lacking, mainly due to the breeding birds of both sexes being highly mobile and absence of territoriality in the males (Schaefer *et al*. 2000). Although males seem to track fertile females (Schaefer *et al*. 2000), we do not know whether the increase in abundance of singing males during second broods reflects an increase in the number of females attempting second clutches. As far as we know at present, only. *c*50% of females start second broods (Schulze-Hagen *et al*. 1999), but this figure may be inaccurate, because most females starting a second clutch move to another area (Wawrzyniak & Sohns 1977, Dyrcz & Zdunek 1993). Still, the concentration of breeding attempts late in the season may be offset by a seasonal decline in offspring quality (Nilsson & Smith 1988, Verhulst & Tinbergen 1991) and in reduced survival prospects of double-brooded females (Bryant 1979, Verhulst & Tinbergen 1991).

### 4.2. Proposals for, and character of, monitoring work

Further intensive monitoring should be conducted of a few selected areas, comprising repetition of the counts throughout the breeding season at regular one-week intervals to detect possible settlement changes in the area. Descriptions, made in parallel with each count, of selected habitat characteristics such as water depth, the vegetation cover would make it possible to define the range of a optimum habitats over the timescale of the whole breeding period.

We propose a long-term monitoring programme of Aquatic Warbler in Poland, censuses being carried out on selected representative plots, within all the present breeding sites. Simultaneous checks (of seasonal changes of plant cover), measurements (of parameters such as plant height; water level, density and structure of plant cover) and descriptions (of characteristic plant species) would make it possible to adapt factors of conservation activity (controlled grazing, controlled mowing, prescribed burning) to changes in the structure of the plant cover. Such monitoring should comprise both optimal and sub-optimal habitats.

Censuses should be made:

1. Every three years on the same plots chosen within all the breeding sites of Aquatic Warbler in Poland
2. Every year on one selected monitoring plot each at the most important breeding sites: western Pomerania, the Biebrza marshes, the Narew marshes and the Lublin Province marshes.

There should be two censuses each season, to coincide with the first and second brood periods of 15-30 May and 15-30 June.

### 4.3. Conclusions

1. Considering that all the main breeding areas of Aquatic Warbler in Poland appear to be identified and that the habitat requirements of the species are well defined, monitoring of Aquatic Warbler should comprise monitoring of changes in habitat characteristics in the most important areas. Clustered distribution of Aquatic Warblers in the occupied areas, intra-seasonal changes in numbers and locally-specific causes of habitat destruction may lead to inaccurate predictions of population trends, unless based on habitat quality assessment. Vegetation sampling should be designed to document the range of ‘suitable areas’.
2. Monitoring should comprise not only optimal breeding habitats but also sub-optimal breeding habitats with low numbers of singing males. Because a proportion of the sub-optimal habitats are small and are dispersed throughout the country, it will be necessary to mobilise local resources for counts and conservation action.
3. In the monitored areas, double counts should be conducted, although in practice this restricts the number of census plots. During second broods, second counts may help to pinpoint the role of some areas, by detecting during the breeding season differential habitat attractiveness and links between fluctuations in habitat features and distribution of birds.
4. Research is needed on the links and relations between subpopulations of the entire meta-population.

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# Large-scale monitoring of the effects of human disturbance on waterbirds: a review and recommendations for survey design

J. A. Robinson and P. A. Cranswick

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Disturbance, especially that caused by human recreational activities, is a threat to waterbirds, particularly since many recreational activities may be increasing in intensity and distribution. Disturbance can have a considerable effect on the numbers of birds using a site and in some circumstances may have consequences for the size of populations. The EC Birds Directive, Ramsar Convention and the African-Eurasian Waterbird Agreement provide the legislative framework for waterbird conservation along the east Atlantic flyway and require signatories to assess the extent, effects and impacts of human disturbance and to mitigate against any deleterious effects. There is also a good scientific basis for monitoring the effects and impacts of disturbance. We recommend that existing large-scale volunteer-based surveys should be used to monitor the extent and distribution of human activities at a variety of spatial scales. However, we argue that these surveys are of little direct use in measuring the effects and impacts of disturbance on waterbirds.

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## 1. Introduction

Wetlands and waterbirds are under intense pressure from anthropogenic activities such as land claim, habitat destruction, pollution, hunting and recreation (Bell & Owen 1990, Ward 1990, Yalden 1992, Tucker & Heath 1994). It is generally agreed that disturbance, especially that caused by recreational activities, is a threat to waterbirds, particularly since many recreational activities may be increasing in intensity and distribution (*e.g.* Ward 1990, Cayford 1993).

It has been estimated that 23 waterbird Species of European Conservation Concern (SPECs) have suffered moderate or large scale declines in the past, due in part, to some form of disturbance (Tucker & Heath 1994). Furthermore, 29% of European sites classified as Important Bird Areas (IBAs) are threatened by the effects of disturbance (Heath & Evans 2000). Although many experimental studies have shown that disturbance, which can be equated to deterioration of habitat, can have a considerable effect on the numbers of individuals using a site, it is generally much less clear how populations of species respond to the stimuli (see Madsen *et al.* 1995 and Hill *et al.* 1997 for reviews).

Effective waterbird protection requires the demonstration and minimisation of the effects and impacts of anthropogenic activity where there is a potential conflict between waterbird conservation and recreation interests. Hill *et al*. (1997) provided a comprehensive set of recommendations for disturbance research that would serve to provide the scientific basis to underpin disturbance research. In contrast, very little attention has focused on the ways in which large-scale volunteer-based surveys can contribute to the monitoring of disturbance at wetlands to fulfil conservation objectives.

In this paper we review the requirements for disturbance monitoring made under the European Community (EC) Birds Directive, the Ramsar Convention and the African Eurasian Waterbird Agreement (AEWA). These three international commitments, in conjunction, provide the legislative framework for the conservation of waterbirds throughout the east Atlantic flyway. We then focus on the scientific basis for monitoring disturbance by reviewing the methods used to measure the effects and impacts of disturbance. Using these reviews as a backdrop, and highlighting the strengths and weaknesses of current 'disturbance' monitoring in the UK by the Wetland Bird Survey (WeBS), we identify the extent to which existing large-scale volunteer-based count schemes potentially can be used to monitor the occurrence and consequences of human activities for waterbirds.

## 2. A legislative framework for monitoring waterbird disturbance

Nations are responsible for implementing various international agreements, directives and conventions that have been introduced to ensure that birds and their habitats are conserved effectively. Along the east Atlantic flyway, the EC Birds Directive 1979, the Ramsar Convention 1971 and the AEWA 1995 provide legislative requirements for disturbance research.

### i. The EC Birds Directive

The EC Directive on the Conservation of Wild Birds (Council Directive 79/409/EEC) provides a legal framework for the conservation of naturally occurring bird species in Europe. Article 2 of the Directive requires the maintenance of populations of bird species 'at a level which corresponds in particular to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements, or to adapt the populations of these species to that level.'

Article 3 requires that Member States should 'take requisite measures to, maintain or re-establish a sufficient diversity and area of habitats for all the species of birds naturally occurring in Europe referred to in Article 1.' Article 4 requires Member States to classify suitable territories in number and size as Special Protection Areas (SPAs). Article 4 specifies that steps should be taken 'to avoid pollution or deterioration of habitats or any disturbances affecting the birds insofar as these would be significant' and that 'outside these protection areas, Member States shall also strive to avoid pollution or deterioration of habitats.'

**ii. The Ramsar Convention on Wetlands of International Importance**

The Ramsar Convention requires signatories to protect wetlands of international importance, to promote wetlands generally and to foster the wise use of wetlands. At least one site in each country must be designated for inclusion in the Ramsar 'List'. With respect to the impacts of human activities, Article 3.1 specifically requires Signatories to 'formulate and implement their planning so as to promote the conservation of wetlands included in the List, and as far as possible the wise use of wetlands in their territory' and, within Article 3.2, '...arrange to be informed at the earliest possible time if the ecological character of any wetland and its territory in the List has changed, is changing or is likely to change as the result of technological developments, pollution or interference.'

Signatories are also required to '...encourage research and the exchange of data and publications regarding wetlands and their flora and fauna.' However, the Convention text is no more specific than this regarding the measurement and monitoring of the effects and impacts of human disturbance.

**iii. AEWA**

The AEWA 1995 requires that Parties should take co-ordinated measures to maintain migratory species in a favourable conservation status, or to restore them to such a status. This Agreement goes slightly further than the Birds Directive and Ramsar; Article III, 2 (e) requiring Signatories to '...investigate the problems posed or are likely to be posed by human activities and attempt to implement remedial actions throughout flyways. Such information can only be collected by long-term schemes which monitor the effects of anthropogenic disturbance on waterbirds.'

For those waterbird populations with particularly unfavourable conservation status, Section 2.1.1 (b) of the Agreement Action Plan requests that Signatories should '...prohibit deliberate disturbance in so far as such disturbance would be significant for the conservation of the population concerned.' Section 4 deals with the management of human activities and in 4.3.6 requests that 'In cases where human disturbance threatens the conservation status of waterbird populations listed in Table 1, Parties should endeavour to take measures to limit the level of threat. Appropriate measures might include, inter alia, the establishment of disturbance-free zones in protected areas where public access is not permitted.'

Section 5 deals specifically with research and monitoring needs and in Part 6 states that Parties '.....shall endeavour to undertake studies on the effects of.......disturbance on the carrying capacity of wetlands used by the populations listed in Table 1 and on the migration patterns of such populations.'

In summary, if nations are to fulfil their commitments under this international legislation, it will be necessary to develop appropriate research that adequately measures:

- The causes, distribution and frequency of potentially disturbing activities nationwide.
- The 'effects' of human disturbance at site-level, especially at protected sites.
- The 'impacts' of human disturbance at the population level, ie what the consequences are of disturbance for the con-

servation status of individual waterbird populations.

## 2. A scientific basis for measuring the effects and impacts of human disturbance

It is important to differentiate between the terms 'effects' and 'impacts' when used in the biological sense. An effect is an observed response, ie a movement of birds (that may only be a temporary displacement) away from a site in response to some stimuli. Furthermore, birds may be able to use alternative sites during periods of high disturbance at the original site without any negative effects on their energy budget. Impacts are of primary conservation importance because they imply a reduction in survival of individuals, which may cause declines in population size. Impacts depend largely on whether alternative sites are available and the energetic costs of displacement (Gill *et al*. 1998). The stages of measuring the effects and impacts of human disturbance are summarised in Fig. 1 (slightly amended from Davidson & Rothwell 1993).

Two approaches have been taken to assess the effects of disturbance on waterbirds. The first method involves recording the distribution of animals before and after disturbance incidents (e.g. Draulans & van Vessem 1985, Bélanger & Bédard 1989, Madsen 1998a). A problem associated with this method is that disruptions to waterbird distribution subsequent to a disturbance event may not have negative consequences because the new distribution pattern may only be temporary; animals returning to their original distribution at a later date to exploit the remaining resources (Owens 1977, Underhill *et al*. 1993). The alternative method is to relate the numbers of animals to the varying rates of disturbance across a number of sites or patches within sites (*e.g*. Tuite *et al*. 1984, Sutherland & Crockford 1992). However, to be able to interpret these data correctly, an assessment of the number of animals using the site in the absence of disturbance is required. Without some

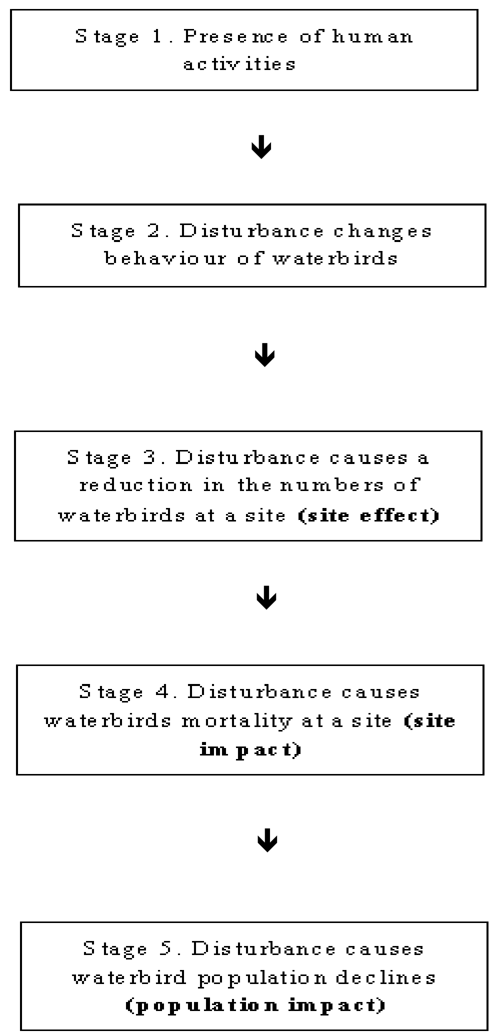


Fig. 1. Stages in measuring the effects and impacts of human disturbance to waterbirds (amended slightly from Davidson & Rothwell 1993).

form of experimental control, the results of these types of studies are flawed. In an attempt to overcome these problems, Madsen (1998b) was able to vary the levels of hunting disturbance experimentally in Denmark and recorded the reactions of waterbirds in terms of displacement and redistribution.

To understand the impacts of disturbance on waterbird populations it is necessary to know not only whether a species avoids sites where humans are present, but also the consequent costs of moving to another site (Gill *et al*. 1998). Gill *et al*. (1996) described a method of quantifying the impacts of disturbance, based on the trade-off between resource use and risk of disturbance. The approach follows a similar technique used to study the effects of predation risk on patch use (Lima & Dill 1990). In effect, waterbirds perceive humans as potential predators. The technique proposed by Gill *et al*. (1996) measures the reduction in the use of a resource in response to disturbance. The approach allows both quantification of the effect of disturbance on numbers at a local scale, and exploration of the potential consequences of changes in disturbance on the size of populations.

Individuals-based population models have focused on the impacts of habitat loss on waterbird populations and provide a conceptual framework for predicting its consequences (Goss-Custard 1985, 1993, Goss-Custard *et al*. 1995a, b, Sutherland 1996a, b, 1998, Pettifor *et al*. 2000). Disturbance can be equated to habitat loss because both factors act to reduce the carrying capacity of a site. In simplistic terms, disturbance and habitat loss give rise to a reduction in food availability leading to movements of birds to other sites and therefore increased density (Goss-Custard 1977 1993, Sutherland & Goss-Custard 1991). Increased density, in turn, results in increased food depletion or competitive interference (or both) so that food intake is affected, reducing the optimality of the habitat and hence its 'carrying capacity' (Goss-Custard *et al*. 1995c, d; Stillman *et al*. 2000). The consequence of this at the metapopulation scale is to increase mortality as birds drop below a critical body mass threshold for survival, leading to flyway-scale population declines as habitat is increasingly lost through disturbance. As habitat is removed or disturbance levels are increased there may be no effects on bird numbers until a threshold density is reached. Beyond this density, density-dependent mortality occurs (Zwarts 1976, Goss-Custard 1977).

In some species, individuals may have to compete strongly to gain access to resources, perhaps because resources are uncommon, are depleted rapidly, because birds are near to the limits of their energy budget, because density is already high or because few suitable alternative sites are available. Therefore, these species are the most likely to be adversely affected by disturbance and habitat loss. Since density-dependent effects operate largely through interference competition between individuals on the feeding grounds in these species, and hence food competition, a method of measuring this density dependence is deemed to be the most appropriate method for estimating parameter values of density-dependent functions.

Density-dependence models can be used to predict the movements and mortality of birds in response to disturbance or habitat loss at a range of spatial scales, from individual-site to global levels. Clearly, the accuracy of such models relies on the accuracy of the parameter values used and therefore intensive studies of the demography, foraging behaviour, intake rates and physiological condition of the waterbirds involved (Goss-Custard 1995c).

## 3. Recommendations for volunteer survey design

**The Wetland Bird Survey (WeBS) - monitoring human activities in the UK**

WeBS is a large-scale volunteer-based scheme that aims to monitor all non-breeding waterbirds in the U.K. At present, WeBS volunteers record a range of human activities at wetland sites and indicate which of these activities are perceived to be disturbing birds. Counts are generally conducted at the weekend, during the morning and, at coastal sites, at high water (Gilbert *et al*. 1988). Many counters stop counting during the summer months when there are few waterbirds at their site. In light of the results of recent review of the human activities data collected by WeBS (Robinson & Pollitt 2002) and the legislative and scientific requirements for measuring the effects and impacts of disturbance on waterbirds, we identify those stages of disturbance measurement (see Fig1) that could be contributed to by volunteer-based count schemes:

**Stage 1**

With the possibility that disturbance may be increasing in many countries in Europe, measuring the extent and distribution of human activities remains a conservation priority. Previous work examining human activities occurring at wetlands in the UK by Davidson *et al*. (1991) and Robinson & Pollitt (2002) has indicated that the ability to monitor the distribution and occurrence of human activities can be of great value in highlighting potentially disturbing activities at sites, without measuring disturbance per se.

In light of the current success of WeBS in collecting these types of data, we suggest that the occurrence of various human activities at individual sites should be monitored through volunteer-based count schemes, which provide a cost-effective monitoring tool that can cover very large areas. There is some evidence to suggest counters are reluctant to provide disturbance data on a regular basis; only 60% of WeBS counters currently provide information on human activities at their sites (Robinson & Pollitt 2002). Therefore, to avoid over-burdening counters, we suggest this type of review should be undertaken periodically using a counter consultation approach, e.g. by questionnaire. A comprehensive list of the types of activities that could be recorded through this consultation is presented in Davidson *et al*. (1993). We also suggest that temporal and spatial variations in the occurrence of potentially disturbing human activities at a site should also be recorded.

This consultation technique would remove the inaccuracy of recording just those activities encountered during single ‘snapshot’ count visits and allow counters to record additional features, e.g. marinas or sailing clubhouses, which are indicative of potentially disturbing activities that may not be recorded on a specific count date but probably occur regularly. Furthermore, human activities that occur during the summer months, when waterbird numbers are low and no counts are made, can also be recorded using this method.

**Stage 2**

As we have explained above, the measurement of behavioural changes within a site as a consequence of disturbance require intensive field studies using an experimental approach, i.e. before-and-after studies using control sites for comparison (Hill *et al*. 1997). In contrast, we suggest that subjective questions, asking counters to measure the perceived 'effects' of human activities on waterbirds, should be avoided. Robinson & Pollitt (2002) suggested that there may be a positive correlation between WeBS counters indicating the presence of disturbance and the degree of disturbance at sites. Furthermore, as mentioned above, 'snapshot' counts, such as those used by WeBS and other volunteer-based count schemes in Europe, do not provide information on human activities occurring at times other than during the count itself. For example, many volunteers prefer to make their counts during late morning when disturbance events may have already caused changes in the behaviour and local distribution of birds.

**Stage 3**

Assessing the long-term effects of disturbance on waterbirds at individual sites is necessary, primarily to ensure that site management is continually sympathetic to the conservation of the waterbirds. It is generally accepted, that site-based work examining the long-term effects of disturbance is best undertaken through more intensive, and scientifically robust methods (*e.g.* following Madsen 1998b).

As mentioned above, WeBS counts are generally undertaken during the weekend, during the early morning and at high water. The potential biases in the data collected as a consequence of these restrictions suggest that such schemes may not be representative of overall human activity patterns across sites. In addition, 'look-see' count methods do not provide an accurate measure of the occurrence or intensity of disturbance at a site. For example, a counter may record numbers of waterbirds on a wetland regularly used for watersports yet, because he arrives there prior to peak activity, his records underestimate levels of human activity at the site. Relationships, or lack of them, between waterbird numbers and levels of human activity measured through volunteer-based surveys are therefore likely to be spurious.

**Stages 4 and 5**

Realistically, it will always remain the responsibility of professional ecologists to measure the impacts of disturbance using predictive population models and resource utilization studies like those suggested by Gill *et al*. (1996). This is largely because of the complexity and rigorous nature of the research required. However, to test the predictions of individuals-based population models there is a need to measure trends in abundance at the population level. To this end, we suggest that survey organisers and population modellers should work more closely together to investigate if and how volunteer-based count schemes could be improved to deliver the data required to test these predictive models.

## 4. Conclusions

In summary, existing volunteer-based count schemes are useful in monitoring the extent and distribution of human activities at sites. In addition, the ability of count schemes to accurately identify worrying trends in the numbers of waterbirds at sites can be of great use in informing sympathetic site management for waterbirds. In the UK, links between downward trends and the effects of potentially damaging activities can prompt statutory regulation of the damaging activity at site-level. However, to demonstrate the effects, let alone the impacts, of such activities requires scientifically robust research rather than existing volunteer-based monitoring. We also suggest that population monitoring through count schemes can be used to test models that predict the likely impacts of disturbance. Predicting the impacts of disturbance accurately will substantially improve our ability to evaluate site-based proposals for potentially disturbing activities rather than having to react to contemporary problems.

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# Incorporating precision, accuracy and alternative sampling designs into a continental monitoring program for colonial waterbirds

M. J. Steinkamp, B. G. Peterjohn and J. L. Keisman

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A comprehensive monitoring program for colonial waterbirds in North America has never existed. At smaller geographic scales, many states and provinces conduct surveys of colonial waterbird populations. Periodic regional surveys are conducted at varying times during the breeding season using a variety of survey methods, which complicates attempts to estimate population trends for most species. The US Geological Survey Patuxent Wildlife Research Center has recently started to coordinate colonial waterbird monitoring efforts throughout North America. A centralized database has been developed with an Internet-based data entry and retrieval page. The extent of existing colonial waterbird surveys has been defined, allowing gaps in coverage to be identified and basic inventories completed where desirable. To enable analyses of comparable data at regional or larger geographic scales, sampling populations through statistically sound sampling designs should supersede obtaining counts at every colony. Standardized breeding season survey techniques have been agreed upon and documented in a monitoring manual. Each survey in the manual has associated with it recommendations for bias estimation, and includes specific instructions on measuring detectability. The methods proposed in the manual are for developing reliable, comparable indices of population size to establish trend information at multiple spatial and temporal scales, but they will not result in robust estimates of total population numbers.

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## 1. Introduction

Colonial waterbirds have been the subjects of extensive scientific studies and have received considerable popular attention for many decades. The millinery trade and other exploitation produced marked population declines of many colonial-nesting birds during the nineteenth century, and initiated the first organized efforts in North American bird conservation at the turn of the twentieth century (Bent 1926). Although the conservation of these species remains an important environmental issue today (Bartle 1991, Brothers 1991, Kushlan 1992, Luthin 1987), it is now realized that the very existence of waterbirds at the boundaries of terrestrial and aquatic ecosystems allows them to serve as important bioindicators of environmental change (Cairns 1987, Custer & Osborn

1977, Kushlan 1993). Hence, knowledge of population trends provides useful information about the effectiveness of conservation activities for these species and the overall health of our ecosystems.

Measuring population change for most colonial waterbirds poses considerable challenges. Their colonies frequently are located in relatively inaccessible locations that preclude access to many nesting pairs, and the number of breeding adults can be enormous, thus preventing accurate counts of individuals. Nesting habitats, nest site preferences, and breeding behavior vary considerably among species and even among populations of a species, and multiple methods are required to survey their populations effectively (Bibby *et al.* 2000, Nettleship 1976, Walsh *et al*. 1995). Crevice-nesting and burrowing species are difficult to survey by any technique (Gaston *et al*. 1988, Savard & Smith 1985), and a better understanding of their population trends will occur only with the development of improved survey methods. Potentially, population surveys measure several parameters including numbers of active nests, numbers of pairs, or the total numbers of adults present at a colony (including both breeding pairs and nonbreeders), producing population estimates that may not be directly comparable.

These potential problems have not discouraged the regular surveys of colonial waterbird populations. The status of individual colonies are routinely monitored for scientific and conservation purposes, while periodic organized efforts are undertaken to estimate population sizes at national and regional geographic scales (Lloyd *et al*. 1993, Sowls *et al*. 1978, Spendelow & Patton 1988). The failure to include estimates of the accuracy and precision associated with the counts complicates comparisons of population change between surveys, resulting in uncertainty concerning the actual extent of population change that occurred over time (Burnham 1981, Johnson 1995, Nichols *et al*. 2000).

In North America, most population surveys of colonial waterbirds have been undertaken at the scale of individual states, provinces or regions (e.g. Erwin 1979, Nesbitt *et al*. 1982, New York Department of Environmental Conservation 1998, Scharf 1998, Scharf & Shugart 1998, Sowls *et al*. 1978). However, the development of a conservation plan for waterbirds has renewed interest in creating a coordinated effort for monitoring the colonial-nesting species at various geographic scales in order to provide population information relevant to the management of these species (Steering Committee 2000). This paper discusses the issues associated with the creation of a coordinated colonial waterbird monitoring program for North America and the need to incorporate measures of accuracy and precision into the survey methodologies to improve the robustness of population estimates for these species.

## 2. Existing Field Methods

### 2.1. Sampling Design

The traditional approach for most colonial waterbird monitoring efforts is to obtain population estimates from every colony within the geographic area of interest (Erwin 1979, Scharf 1998, Scharf & Shugart 1998, Shuford & Ryan 2000, Texas Colonial Waterbird Society 1982). This approach reflects the temporal and geographic shifts in colony locations and the changes in species composition and population sizes that normally occur over time, and the belief that comparable data are most likely to be obtained only by surveying every known breeding location.

At larger geographic scales, this approach requires considerable coordination and expenditure of resources in order to be implemented in the field, not only for the population surveys but also for colony inventories required to locate newly created colonies and colony sites that may have shifted following previous surveys. This need for substantial resources to implement regional colonial waterbird surveys usually allows these surveys to occur only at intervals of 5-10 years or longer, eliminating the chance of detecting short-term changes in most populations and the ability to implement appropriate conservation and management activities in the event of rapid short-term population declines (Fig. 1).

### 2.2. Survey Methods

During these population surveys, survey methods tend to be standardized in an attempt to reduce variability in the estimates of population size. The assumption is that with the use of standardized methods, changes in counts between surveys reflect actual changes in population size and not changes in the proportion of the populations that were actually detected by the method. Many factors can influence the detection probabilities associated with a survey technique (Jolly & Dickson 1983, Nichols *et al*. 2000), and unless detection probabilities are explicitly measured during the surveys, the changes in counts between surveys may reflect changes in population size, detection probabilities, or some unknown combination of both.

Standardization of survey methods also implies that a single technique is equally appropriate for all nesting habitats and locations occupied by a species. This assumption may be true for some species with specific breeding habitat requirements. However, widely distributed species frequently occupy a variety of nesting habitats, and a single survey technique may have different detection probabilities in each habitat. Hence, temporal shifts in colony locations may be accompanied by changes in detection probabilities and confound analyses of population change between colonial waterbird surveys.

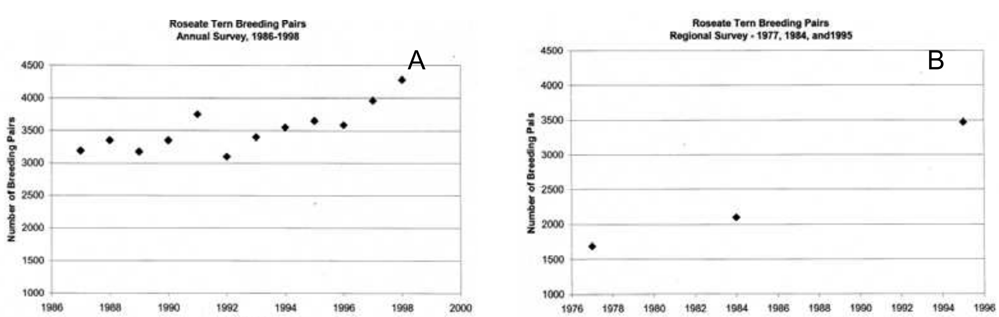


Fig. 1. A comparison of the difference in interpretation of Roseate Tern *Sterna dougallii* data collected (a) annually, and collected (b) every 5-10 years. Data collected annually shows fluctuations in numbers of breeding pairs each year, whereas data collected by regional surveys conducted every 5-10 years may be interpreted as increasing numbers of breeding pairs.

Observer variability is well known to influence counts obtained during bird population surveys (Erwin 1982, Prater 1979, Verner 1985). In order to reduce this variability, it is customary to use either a single observer or a small number of observers to conduct colonial waterbird surveys. These observers receive training in survey methodology and in the estimation of large concentrations of birds. While standardization improves the consistency of data collect within each survey, temporal changes in observers and comparisons among observers surveying over large geographic areas can still result in substantial variability in population estimates obtained during colonial waterbird surveys (Gibbs *et al*. 1988).

### 2.3. Survey Timing

When colonial waterbirds are surveyed over large geographic areas, the use of a small number of observers necessitates conducting surveys throughout the breeding season. Attendance rates are known to vary with the stage of breeding chronology (Hatch & Hatch 1989, Jones 1992, Piatt *et al*. 1990, Rothery *et al*. 1988), and changes in attendance rates can be confounded with population change at a colony. Nest failure may result in intercolony movements of adults during a nesting season (Massey & Atwood 1981), and these movements could result in the double counting of adults within a single survey period. All of these factors contribute to increased variability in the estimates of population size obtained during these surveys, and reduces the benefits obtained by standardizing observers.

## 3. Proposed Monitoring Program

Implementing a colonial waterbird monitoring program across North America poses considerable challenges. Limited resources will preclude any attempt to obtain population estimates at every colony, and sampling populations of most species will be a necessity. Even sampling colonies will require a large number of observers to conduct such surveys over North America. Multiple methods will likely be employed to survey most species. Estimation of detection probabilities associated with each observer-method combination is essential to produce comparable population estimates over time. Taking these factors into consideration, the proposed North American colonial waterbird monitoring program is outlined below.

### 3.1. Inventory

Inventories conducted at the scale of states and provinces will provide information on the current species composition, size, and distribution of waterbird colonies. Information obtained from these inventories will be used to develop an appropriate sampling framework for the widely distributed species. These inventories will be updated periodically to permit adjustments in the sampling design to accommodate temporal changes in distribution and abundance for each species.

### 3.2. Sampling Design

For species with small, locally distributed breeding populations, every colony will be surveyed in order to develop population estimates. Examples include populations of Elegant Terns *Sterna elegans* in southern California and Sooty Terns *S. fuscata* and Brown Noddies *Anous stolidus*) on the Dry Tortugas of Florida as well as endangered or threatened species such as Roseate Tern *S. dougallii*.

Most species are more widely distributed and a sample of their colony sites will be regularly monitored to estimate changes in population size. For some species, the appropriate sampling scheme will be developed based upon their patterns of distribution and abundance across their entire ranges. This approach is appropriate for nomadic species such as White-faced Ibis *Plegadis chihi* and White Ibis *Eudocimus albus* and for species with regional distribution patterns such as Wood Storks *Mycteria americana* in the southeastern United States and Ashy Storm-Petrel *Oceanodroma homochroa*, Xantus' Murrelets *Synthliboramphus hypoleucus* and other species found only along portions of the Pacific Coast. For the most widely distributed species, appropriate regional sampling schemes will be developed and range-wide population estimates produced through the summation of regional estimates. Dual-frame sampling, which accommodates for the bias of known nest sites (Haines & Pollock 1998), will be one sampling design considered and tested for its efficacy in sampling colonial waterbirds.

### 3.3. Method Development

Survey techniques remain poorly developed or nonexistent for some groups of species such as crevice-nesting alcids and nocturnal birds. A high priority is the development of appropriate survey methods with their associated detection probabilities for these taxa. Use of new technologies, such as high-frequency surveillance radar (Burger 1997) and modifications to existing methods will be encouraged as long as these methods allow for the determination of detection probabilities. New and improved survey techniques will be incorporated into the monitoring efforts once their reliability has been established, assuming that the necessary resources are available to support their use.

A handbook of recommended methods is under development for this monitoring program (Steinkamp & Peterjohn 2000), recognizing that several methods may be needed in order to monitor adequately all populations of a species. Approaches for determining detection probabilities are described for each method in order to produce estimates of precision for every population. Methods lacking sufficient levels of accuracy or the inability to estimate detection probabilities have been excluded from this handbook.

Instead of recommending a single method for surveying each species, multiple methods are described for use in various habitats and generally follow well-established techniques for surveys of colonial waterbird populations (Bibby *et al*. 2000, Nettleship 1976, Walsh *et al*. 1995). The advantages and disadvantages of each method are discussed thus allowing the user to decide which technique is most appropriate (given the available resources) to survey specific colonies. Determination of detection probabilities is a critical component of these methods, for it allows for the comparison of results between colonies within a region and between years at each location.

This handbook also provides recommendations on sampling within colonies as an alternative to attempting a complete count of breeding adults or nests. For any colony, the long-term availability of resources influences such a decision; the handbook recommends the consistent use of one approach or the other.

While several techniques for sampling within a colony have been developed (Anker-Nilssen & Rostad 1993, Bibby *et al*. 2000, Nettleship 1976), their suitability for detecting changes in population levels requires additional study. Population change within most colonies does not tend to be a series of random events, but is normally most evident at the periphery of colonies while preferred nest sites in the center of colonies tend to be consistently occupied. Hence, a simple random placement of sample plots within a colony may not accurately represent the population changes occurring over time (Walsh *et al*. 1995). Accessibility also influences the placement of sample plots; topography may determine the placement of plots at cliffs or other inaccessible locations.

**3.4. Survey Timing**

Because sampling allows fewer colonies to be regularly monitored, the surveys can be more concentrated at the most appropriate stage of the breeding chronology, so that the colony attendance of each species is represented accurately (Byrd *et al*. 1983, Hatch & Hatch 1988, 1989, Jones 1992), the intent being to conduct surveys at the same nesting stage each year. The survey timing will vary from locality to locality as a reflection of geographic differences in breeding chronologies, and may also have to vary annually at any colony to reflect between-year differences in the timing of breeding activities. For Least Terns *Sterna antillarum* and other species whose breeding adults frequently move between colonies during a breeding season, survey timing will have to be coordinated at a regional level to avoid the potential for multiple counts of adults.

**3.5. Demographic and Habitat Monitoring**

Population indices provide the resource manager with only one piece of necessary information. To make scientifically informed decisions on population and habitat management, managers must have demographic information on populations, including survival measures, and site-specific habitat information. This monitoring program proposes to identify reference sites within regions where demographic information will be collected. Site selection will be dependent on regional sampling designs and priority species. A standardized habitat collection protocol will be developed and implemented at colony sites.

**3.6. Observers**

This monitoring program will require the cooperation of numerous professional biologists and volunteer birdwatchers. Observer training in survey methodologies and estimation of large numbers of birds is essential to reduce some of the variability associated with the implementation of a continental monitoring program (Bibby *et al*. 2000, Erwin 1982). The determination of observer-based detection probabilities will also improve the comparability of data collected during these surveys.

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**Ornis Hungarica 12-13: 217-222. 2003**

# Hierarchical models and the analysis of bird survey information

J. R. Sauer and W. A. Link

Sauer, J. R. and Link, W. A. 2003. Hierarchical models and the analysis of bird survey information. – Ornis Hung. 12-13: 217-222.

Management of birds often requires analysis of collections of estimates. We describe a hierarchical modeling approach to the analysis of these data, in which parameters associated with the individual species estimates are treated as random variables, and probability statements are made about the species parameters conditioned on the data. A Markov-Chain Monte Carlo (MCMC) procedure is used to fit the hierarchical model. This approach is computer intensive, and is based upon simulation. MCMC allows for estimation both of parameters and of derived statistics. To illustrate the application of this method, we use the case in which we are interested in attributes of a collection of estimates of population change. Using data for 28 species of grassland-breeding birds from the North American Breeding Bird Survey, we estimate the number of species with increasing populations, provide precision-adjusted rankings of species trends, and describe a measure of population stability as the probability that the trend for a species is within a certain interval. Hierarchical models can be applied to a variety of bird survey applications, and we are investigating their use in estimation of population change from survey data.

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## 1. Introduction

In studies of the population ecology of birds, collections of species are often the subject of analyses. In North America, population change and abundance of groups of species that share a common characteristic such as migration status (*e.g.* Neotropical migrants) or breeding habitats (*e.g.* grassland birds) have been the subject of much public interest due to perceived declines (*e.g.* Peterjohn & Sauer 1999, Robbins *et al*. 1989). Furthermore, conservation initiatives focus on taxa such as waterfowl (*e.g.* the North American Waterfowl Management Plan), shorebirds, or colonial waterbirds that share a common life history characteristic. Managers are interested in summarizing information for these taxa, and may use the collective response of all species as a response measure for management.

A common summary of group population change, or an interval-specific trend, is often of primary interest to managers. The average population trend, number of "declining" species, number of species with significant trend estimates, ranks of species by population trend, and lists of species with extreme changes are often presented for groups (Link & Sauer 1995). Managers also seek to identify species whose populations are unstable or changing. In this paper, we discuss some of the difficulties associated with summary of attributes of groups of species. Our focus is on summary of collections of estimates of population trend, but the hierarchical model approach we describe can be used for many attributes. A more complete discussion of the models and methods used in this paper is available in Sauer and Link (2002).

**1.1 Are Summaries Meaningful?**

The fundamental notion associated with summary of group attributes is that the collection of species is meaningful, in that some common characteristic of the species permits summary of population attributes among species. Often, conceptual and practical difficulties exist with any summary. Practical concerns include the problem that quality of information may vary greatly among species; simple averages and summaries of estimates can be misleading. Conceptual difficulties are based on the notion that all species are different, and that all groupings rely on the characteristic of interest to be a common influence for all groups. For example, several North American grassland-breeding birds also winter in South America, while others winter in North America. Any grouping is likely to compartmentalize only a portion of the variation associated with the attribute of interest (Mannan *et al*. 1984). It is also likely that taxonomic similarity is partially confounded with other possible groups, and hence any similarities may reflect common ancestry (Pagel & Harvey 1991).

**1.2. Difficulties Associated with Imprecise Information.**

Collections of estimates of population trend tend to differ in both estimated magnitude and precision of individual estimates. An imprecise trend estimate may be quite large while still having a confidence interval large enough to include zero, indicating that the trend is not significantly different from zero. Magnitude alone is therefore not sufficient to establish the importance of a trend estimate. However, 'statistical significance' is also a flawed criterion, because a very small rate of change may be identified as 'statistically significant' but be of no practical significance. Separating notions of statistical significance from magnitude of trend has been a conceptual difficulty for exercises in species prioritization. In any collection of trend estimates, some are very imprecise, some are very precise, and all summaries of results are influenced by these differences in estimated precision. The consequences are:

1. Simple averages of trend estimates are generally not good descriptors of the collection.
2. Ranked lists of trends do not reflect the real ranking of trends.
3. Number of species with positive trend estimates is not a good estimate of the number of species with positive trends.

**1.3. Analysis of Collections of Estimates**

For single species analyses, we generally consider the data $Y_s$ to be governed by a fixed, unknown parameter $\theta_s$. Statistics are based on distribution of data, given the unknown parameter, $f(Y_s|\theta_s)$. For multiple species, it is reasonable to view parameters as random variables sampled from some distribution. These multi-level models in which data and parameters are both random variables are called hierarchical Models. In these models data are observed, but their distributions are described conditionally on realized values of parameters that are also random variables. This necessitates that one or more additional levels of distributional assumptions should be included in the analysis.

Hierarchical models are often analyzed using Bayesian methods. Bayes approaches are model-based, and are used to make probability statements about $\theta_s$ (Gilks *et al*. 1996). In a Bayes analysis, we define the standard sampling distribution of the data given the unknown parameter, or $f(Y_s|\theta_s)$, but also define the distribution of the parameters in a prior, or $\pi(\theta_s|\Psi)$, where $\Psi$ is a represents hyperparameters that govern the distribution of the parameters. Bayesian inference about the $\theta_s$ is based on the posterior distribution $f(\theta_s|Y_s)$. Unfortunately, derivation of the posterior distribution is often difficult mathematically, limiting the use of Bayes analyses for complicated models. Definitions of the prior also can be controversial, because the prior makes assumptions about the distribution of parameters.

## 2. Implementing Hierarchical Models

In earlier publications, we used empirical Bayes methods to implement a hierarchical model for species group attributes (Link & Sauer 1995). In empirical Bayes, hyperparameters are estimated using information from the data. We applied a simple model to the case of estimation of a prior mean trend for the group, and then we estimated posterior means of the species trend parameters. These 'shrunken' estimates are a weighted average of the prior mean (the estimate of the group mean trend parameter) and the estimated trend. The resulting estimate for each species is intermediate between the original estimated trend and the prior mean, with the actual value dependant on the relative precision of the original trend. We used this model to re-order trends (Link & Sauer 1996), and to estimate the number of increasing species using a bootstrapping procedure (Link & Sauer 1995). However this approach is limited, because quite simple models must be used.

Here, we implement a complete analysis using Markov Chain Monte Carlo (MCMC) methods, a very flexible procedure for fitting hierarchical models. MCMC is a simulation-based approach to estimation, in which:

1. A model is defined in terms of distributions of parameters and hyperparameters.
2. The distribution information for each variable is written as 'full conditionals,' distributions with all other parameters being fixed.
3. An iterative sampling is conducted using these full conditionals.

This iterative procedure produces results that converge on posterior distributions for the parameters. See Spiegelhalter *et al.* (1995) or Sauer and Link (2002) for more details of the estimation procedure.

## 3. Our Model

We assume that a series of *n* trend estimates $\hat{\beta}_s$, $s$=1, 2,..., $n$, exist, and that these estimates are normal random variables with parameters $\beta_s$ and $\sigma^2_s$. The estimated variance $\hat{\sigma}^2_s$ is distributed as a chi-square. The parameter $\beta_s$ is distributed as a normal distribution with hyperparameters $\mu$ and

$\tau^2$. The hyperparameters are also assumed to follow distributions, with $\mu$ distributed normally (mean 0, variance=100 000), and $\tau^2$ and $\sigma^2_s$ are assumed to follow gamma distributions.

To fit this model, we used Program BUGS (Spiegelhalter *et al*. 1995). In this program, a simulation is conducted as described above. After a large number of iterations, the results converge on posterior distributions. After this convergence occurs, the simulation is continued and each iteration provides a set of replicate results based on sampling from the distributions. Consequently, means and variances from the simulation results can be used as estimates of parameters and hyperparameters and their variances.

**3.1 Estimates Produced by our Analysis**

The estimates associated with parameter $\beta_s$ represent the posterior mean trend estimates for individual species. These can be thought of as precision-adjusted estimates that are 'shrunken' toward the overall prior mean estimate ($\mu$). These numbers are similar to the re-ordered trend estimates described by Link & Sauer (1995), but the MCMC approach better accommodates imprecision in the estimates of precision than did the empirical Bayes approach. The number of species with positive trend estimates ($N_{Inc}$) are estimated directly from the MCMC results, simply by counting the number of positive posterior mean trend estimates from each MCMC replicate and using these as replicates to obtain a mean and variance.

**3.2 Defining Population Stability**

Another attribute that can be derived from the MCMC analysis is a notion of stable populations. Population stability has proven difficult to define using estimated population trends because of the difficulties associated with use of magnitude of trends, because large estimated trends may simply be poorly estimated. However, use of the posterior mean trends from the MCMC accommodates the relative imprecision in the context of the collection of estimates, and can be used to define a stable population. We define stability in terms of a maximum acceptable deviation of trend from 0, denoted as $\delta$. The probability that population is stable can be defined as $\Pr(\beta_s \in (-\delta,\delta)|Y)$, that is, the probability that the posterior mean is in the interval, conditional on the observed data Y. Given $\delta$ and a probability *p*, we can estimate the probability that $\beta_s$ is not in $(-\delta,\delta)$ exceeds *p*, or $\Pr(\beta_s \in (-\delta,\delta)|Y) \leq 1-p$. This quantity can be evaluated directly from the MCMC replicates, simply by determining whether each replicate of $\beta_s$ is in the interval $(-\delta,\delta)$. The proportion of replicates that fall outside the interval is an estimate of *p*.

**3.3. MCMC Analysis**

The North American Breeding Bird Survey (BBS) provides population change data for 28 species of grassland-breeding birds for the survey interval 1966-2000. The BBS is a roadside survey, conducted along secondary roads in the United States and Canada. The 24.5-mile routes are surveyed once each year, in June, and are composed of 50 stops, at which 3-minute point counts are conducted. Total counts of individuals of each species summed over the route comprise the yearly index to abundance. Estimating trends (% change /

year) over the survey interval using the Route-regression trend estimates (Link & Sauer 1994) provides estimates that differ greatly in precision and magnitude among species, although many species are declining. 61% of species have significant negative trends, and only 18% of species have trend estimates >0.

From the North American Breeding Bird Survey, we conducted the MCMC analysis on the 28 grassland bird species for which trends could be estimated over the interval 1966-2000. We calculated the posterior mean estimates $\beta_s$, estimated the number of species with positive trend estimates, and calculated the probability that each species is stable for $\delta$=2%/year.

Species rankings are summarized in Fig. 1, in which the trends are ranked by size of posterior mean and the estimated trends are displayed for each posterior mean estimate. The posterior mean estimates show less variation, especially for the species with extreme estimates of increases and declines. This is evident from observation of individual species estimates. For example, the estimated trend for Henslow's Sparrow *Ammodramus henslowii* was -7.46%/*yr*, n=155, but the posterior mean trend was

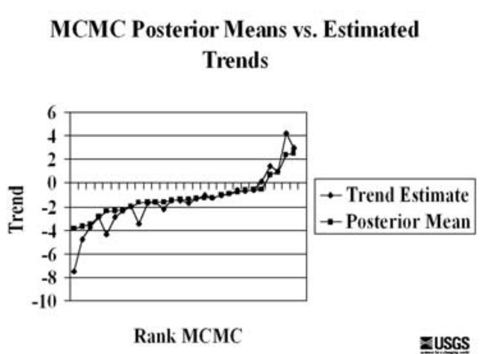


Fig. 1. Posterior mean trends, ranked by magnitude, displayed with estimated trends.

-3.91%/*yr*. For Baird's Sparrow *A. bairdii* the estimated trend was -2.88, n=124, but the posterior mean was -2.35. The populations of four species were estimated to be unstable, using the 2% criterion and a critical value of $P$<0.10: Eastern Meadowlark *Sturnella magna*, posterior mean trend -2.87, $P$<0.001; Grasshopper Sparrow *A. savannarum*, -3.71, $P$<0.001; Henslow's Sparrow and Sprague's Pipit *Anthus spragueii*, -4.73, $P$=0.055. The $N$ of species with positive trend estimates was 5.16 species (*SE*: 1.216), while the naive estimate (*i.e.* based on the estimated trends) was 5. Although similar, the MCMC estimate has a precision estimate associated with it. In this case, the similarity indicates that most species in the group were quite precisely estimated.

## 4. Benefits of Hierarchical Models

The hierarchical models described here provide an appropriate conceptual framework for dealing with collections of estimates. The hierarchical structure provides a framework for estimating attributes associated with the parameters, and the MCMC provides a convenient tool for estimation. The derived attributes such as population stability and number of increasing species can be conveniently estimated during the MCMC simulations. Although some of these attributes can also be defined using empirical Bayes methods, the MCMC approach is superior in that it provides much greater flexibility in defining models and implementing the estimation. In the case described here, the MCMC had an additional component that accommodated uncertainty in estimation of the variance of the estimated trends, and our earlier empirical Bayes procedures could not accommodate this source of variation.

The population stability metric is a useful attribute, in that it resolves the consistent difficulty associated with defining population stability based on magnitude of estimated trends. Because estimated trends differ greatly in precision, the magnitude of the trend does not convey the significance of the trend. Use of the posterior mean estimates does accommodate the relative precision of the estimates and is an appropriate measure of actual magnitude of trend.

We note that there are many other applications for hierarchical models in estimation of population attributes. We are developing methods for estimation of population change using hierarchical models that will accommodate regional variation in precision of time series (Link & Sauer 2002). Finally, we note that although hierarchical models are computer intensive, they are now relatively easy to implement in programs such as BUGS.

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**Ornis Hungarica 12-13: 223-228. 2003**

# Which method is most suitable for censusing breeding populations of red-backed (*Lanius collurio*) and great grey (*L. excubitor*) shrikes?

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S. Kuźniak and G. Lorek

Tryjanowski, P., Hromada, M., Antczak, M., Grzybek, J., Kuźniak, S. and Lorek, G. 2003. Which method is most suitable for censusing breeding populations of red-backed (*Lanius collurio*) and great grey (*L. excubitor*) shrikes? – Ornis Hung. 12-13: 223-228.

Up to now, studies have shown that it is very difficult to estimate accurate numbers of shrike (*Lanius spp*) breeding pairs. During our shrike biology research in Western Poland this problem was evident. Actual breeding pair numbers were derived from a combination of special counts in the pre-breeding period, intensive nest searches and colour ringing. Empirical tests have shown point counts and line transect methods, with regard to Red-backed (*Lanius collurio*) and Great Grey (*L. excubitor*) Shrike population size estimates, to underestimate numbers severely. Furthermore, we have found that the 'improved mapping technique' underestimates the actual numbers of Red-Backed Shrike breeding territories by as much as 45-80%. It underestimates the actual numbers for the Great Grey Shrike by just under 40%. In the case of the Red-Backed Shrike, the accuracy of results from the 'improved mapping technique' was negatively correlated with the population density. However, we did manage to achieve accurate numbers of shrike breeding pairs, but only by using specific methods. For the Great Grey Shrike, counts during pair formation period yielded the most precise results and for the Red-backed Shrike, intensive nest searches proved to be the most effective method.

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## 1. Introduction

The abundance estimates of any bird species are potentially useful for faunistic, zoogeographic or long-term monitoring studies. Information covering longer time periods is crucial for efficient bird conservation. One of the fundamental conditions for such studies is the use species density estimation methods that are valid, applicable and reliable, particularly where endangered species are concerned. Accurate assessments of population size and of the changes it undergoes in time and space form the basis of good species management and sound conservation projects (Marzluf & Sallabanks 1998).

In recent decades, a documented widespread decline has been recorded for all European shrike species over most of their various ranges (Yosef 1994, Bauer & Berthold 1996). The Red-backed Shrike and Great Grey Shrike have the status of declining species in Europe (Tucker *et al*. 1994). Here, we present results of our comparison of the main census methods for these two shrike species. The data arising from this study originated from research into several aspects of the species' biology and ecology in the Wielkoposka and Śląsk regions (Kuźniak 1991, 2000a; Kuźniak *et al*. 1995, Tryjanowski *et al*. 1999, Kuźniak & Tryjanowski 2000, Lorek 1995a, 1995b; Lorek *et al*. 2000).

The main aim of our research was to ascertain whether the methods used for shrike censusing are suitable for monitoring studies, and in particular, whether the results they provide are reliable.

## 2. Study area, material and methods

The study was carried out in the Wielkopolska region, over the area within a radius of 100 km of Poznań (52°28'N; 16°48'E). The study plots were located in a typical agricultural landscape comprising arable fields, meadows and small woodlots. The study areas (sizes in km$^2$) were located near the towns of Odolanów (160 km$^2$) and Koło (176 km$^2$) (Great Grey Shrike censuses), and near the towns of Leszno (10 km$^2$) and Koło (12 km$^2$) (Red-backed Shrike censuses) (see Tryjanowski *et al.* 1999 and Kuźniak & Tryjanowski 2000 for more details about shrike habitats and habitat selection). The Great Grey Shrikes were censused from March through June 2000, but earlier (1990-1999) we had studied in detail Great Grey Shrike breeding ecology not only on the same plots but also in other parts of the Wielkopolska region (Tryjanowski *et al*. 1999). The territories were classified as occupied on the basis of observed territorial behaviour and nests found. The observed bird locations were recorded on maps and notes were taken on territorial behaviour, hunting strategy, habitat selection and (especially) the presence of nests. During the breeding season, a total of 24 census days was spent in the field in the 100 km area.

The Red-Backed Shrikes were censused from May through July 1999 and 2000 on the Leszno study plot and in 2000 on the Koło study plot. Particular attention was paid to the location of all nests. The history of each nest was recorded. Furthermore, the adult birds were trapped and colour ringed to aid subsequent intensive observation. During the breeding season, a total of 30 research days was spent the field in the 10 km$^2$ area.

We simulated a situation typical of a census using the 'improved mapping technique' so that we could compare several census methods (Tomiałojć 1980). Two controls were chosen for the Great Grey Shrike (in April-May) and four (May-June) for Red-backed Shrike. The efficiency of the 'improved mapping technique' was estimated on the basis of a comparison with the data obtained by more sophisticated methods that are assumed to give 100% breeding pairs nesting on the research plot. Point counts and transects were made twice, the first in the second half of April and second in the second half of May in 2000 in the Odolanów and Koło Great Grey Shrike study plots and the Koło Red-backed Shrike study plot. The data were analysed using the SPSS statistical package (Norusis 1986).

## 3. Results

### 3.1. Actual density of breeding pairs

In 2000, 30 breeding pairs (bp) of the Great Grey Shrike were found in the

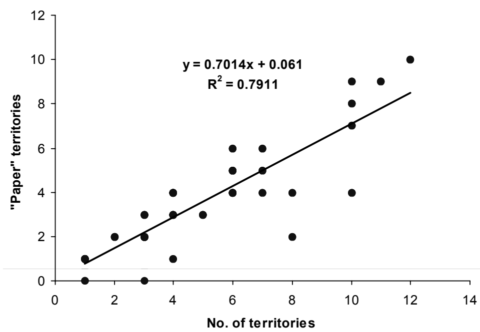


Fig. 1. The relationship between the true number of territories and the number of 'paper' territories, established on the basis of the results from mapping ($p<0.0001$).

Odolanów study plot and 29 in the Koło study plot, giving an average density of 17.63 bp/100 km$^2$. From 1998-2000, we found 47, 52 and 58 bp of Red-backed Shrike respectively, in the Leszno study plot and in 2000, 64 bp in the Koło study plot, giving an average density of 4.5 bp/km$^2$.

### 3.2. Efficiency of count point and transect methods

11 bp out of 59 bp (19%) of Great Grey Shrike in both study plots were detected by point count, the detectability not differing between the study plots ($\chi^2$ with Yates correction=0.01, p=0.92). The transect method yielded 10 bp of Great Grey Shrike (17%), the detectability between plots showing no differences ($\chi^2$ with Yates correction=0.04, p=0.85).

### 3.3. Efficiency of the mapping method

In 2000, 24 bp out of 59 bp (41%) of Great Grey Shrike were found by the 'improved mapping technique' in both study plots. We found no significant differences in the detection rate between plots ($\chi^2$ with Yates correction=0.03, p=0.86). From 1998-2000, the number of territories of the Red-backed Shrike detected by the 'improved mapping technique was significantly correlated to the actual number of nesting pairs within 1 km$^2$ squares ($p<0.0001$). However, only 74±27% of nesting pairs are detected by mapping method (range 0-100).

### 3.4. The data from regional studies vs. actual density

Density of Great Grey Shrike bp in the large study plot in the Wielkopolska region was estimated as 1.6-4.3 bp/100 km$^2$ in the regional bird monograph (Kuźniak 2000b). Kuźniak located 7 bp near Odolanów, but only two near Koło, a value at least seven times lower than that arising from our more comprehensive methodology. The numbers of Red-backed Shrike were estimated only in width range level (Kuźniak 2000a) as a common breeding species whose breeding density varied from 0.1-6.4 bp/km$^2$); our results generally support this value, but suggest that the real value is closer to higher numbers obtained in previous estimates.

## 4. Discussion

The central-western part of Poland, the Wielkopolska region, boasts one of the longest-running farming traditions in Poland. Most of the region recently has been affected by the introduction of chemical and mechanical cultivation (Denisiuk *et al.* 1992), and its development stage now quite representative of that in Western Europe. The processes now taking place in agro-ecosystems in this region are probably indicative of future changes in Polish agriculture generally. However, despite intensive agriculture occurring over a relatively long period, one of highest breeding densities of Great Grey Shrike has been noted in this region (Kuźniak *et al*. 1995, Lorek 1995a, Tryjanowski *et al*. 1999). Similarly, breeding densities of Red-backed Shrike observed in 1999 to 2000 (*c*5 bp/km$^2$) rank among the highest in Europe (obtained on plots similar in size; likewise, they contained other than preferred habitats; Glutz von Blotzheim & Bauer 1993, Dombrowski *et al*. 2000). There is the caveat that censuses performed in conditions of high densities can produce results subject to greater error than when other conditions apply (DeSante 1981, Tomiałojć & Lontkowski 1989), but on the other hand, results of censuses on high-density areas can be more reliable in reflecting the species' population abundance.

First of all, it is important to differentiate between the main methods of shrike censuses (point counting, transect and the 'improved mapping technique') and to suggest why their use has produced underestimates. As we have shown, the first two methods are markedly less effective than ours; our results corresponded to those reported by other authors (*e.g.* Tomiałojć 1987, Tomiałojć & Verner 1990, Surmacki & Tryjanowski 1997). Even the third method we have shown to be unsatisfactory, especially in the case of Great Grey Shrike.

**4.1. Why should shrike censuses have been so ineffective?**

We suggest that above all, these anomalous results arise from specifics of shrike biology and from the periods selected for observations; historically, the general schedule of shrike censusing has in the past been adjusted to that pertaining to the rest of the bird community. Normally, study censuses start in April, but the peak of Great Grey Shrike breeding activity occurs half in March. During this early part of breeding activity, both members of the pair build the nest and the main body of pair and social interactions takes place (mate guarding, territory defence, copulation, courtship feeding) (Lorek 1995b). Also, the most intense song activity is practically confined to this brief time period, which makes discovery of the birds much easier. From early April, often until about the 20th, the birds exhibit shy and skulking behaviour, because this prevents attention being drawn to females incubating, or being fed by males, or to changeovers at the nest. By then, nest and bird location becomes increasingly confounded by the rapid foliation of trees and bushes

In the period between arrival from return migration and nest building, the Red-backed Shrike remains very silent, except when the males sing to establish territory, or fight other males to defend it; direct observation of these activities is the best method of detecting this species (Durango 1956). A little later in the breeding season, it is even more difficult to confirm the presence of birds, and then it is mainly through more desultory singing. Activity then increases in the period of intense nestling feeding (late June), when usually standard observations of breeding bird communities have ended.

**4.2. How do we improve shrike censuses?**

***1. The choice of a study plot.***

The ideal study area should cover *c*50-100 km² for the Great Grey Shrike and *c*5-10 km² for the Red-backed Shrike in conditions and circumstances similar to those in western Poland. In both cases, it would be possible for one person to perform the censuses. Before beginning a census, it is important to assess potential shrike habitats (trees, bushes) using maps, photos and field reconnaissance, and to record them particularly thoroughly. It is recommended that possible shrike habitats be identified beforehand in late autumn, when searches for nests from the previous breeding season should be undertaken (Seitz 1992). The nests of each species have a characteristic construction and often contain food remains, mainly large insects and grasshoppers.

***2. The timing of study.***

The optimum date for census is from 15 March to 30 April when 2-3 controls of the entire plot should be made, followed by 1-2 controls between 15 May and 15 June. It is necessary to complement visual records by searching for nests. The Red-backed Shrike may nest at in high densities in some habitats, such that the distance between nests of different pairs may be as little as 40 m (mean 225 m, Kuźniak & Tryjanowski 2000, also their unpublished data).

***3. More attention to be paid to specifics of shrike biology.***

Visual observations should be complemented by searches for pellets, food remnants, prey, bird plucks and, above all, impaled caches. Birds frequently use isolated or free-standing *Prunus*, *Rosa* or *Crataegus* bushes and barbed-wire fences. These places may be favoured for more than one season and so the quantities of droppings, pellets and another remains can be significant. Of course, records of pellets or prey remnants alone cannot be the basis of territorial pair confirmation, but it is a sign that a more intensive search for the birds themselves and their nests might be profitable. Our study is the first-large scale comparison of the commonly used method for shrike censusing. Despite large area censuses was carried out by us, our results cannot be regarded as exhaustive in detecting all methodological problems with shrike census work. More comparative studies of methods, especially in areas with low numbers of Great Grey and Red-backed Shrikes, are urgently needed.

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**Ornis Hungarica 12-13: 229-236. 2003**

# On the importance of nestbox age in monitoring populations of small hole-nesting birds

I. Vilka

Vilka, I. 2003. On the importance of nestbox age in monitoring populations of small hole-nesting birds. – Ornis Hung. 12-13: 229-236.

In Latvia, monitoring of small hole-nesting birds that use nestboxes has been carried out during a 20-year period (1981-2000). For data analysis, 20 nestbox plots from different parts of Latvia were used. The total number of nestboxes was 2140. The main species breeding in the nestboxes were Pied Flycatcher *Ficedula hypoleuca* and Great Tit *Parus major*. Breeding populations of both species declined during the study period. These negative trends may reflect the actual state of the populations or may have been caused by other factors, such as nestbox aging. That birds tend to avoid old nest boxes has been mentioned in the literature, but many nestbox surveys have not taken this factor into account. To test the importance of nestbox age, the current study investigated nestbox occupancy against nestbox age. It found a significant correlation between declining occupancy rate and nestbox age. When in one study plot 7-year old nestboxes were replaced by new boxes, in the subsequent year the number of the Pied Flycatcher broods substantially increased. However, Pied Flycatcher clutch size increased in older nestboxes.

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## 1. Introduction

The tendency for birds to avoid selecting as nest sites nestboxes several years old (in comparison with new nestboxes) but still in good condition has been observed in several nestbox surveys in Europe (Shcherbakov 1956, Blagosklonov 1970, Lundberg & Alatalo 1992) and in a survey in Latvia (Mihelsons 1958a). Some possible explanations for nestbox aging affecting their selection by hole-nesting birds have also been proposed (*e.g.* microclimatic conditions, visibility, the presence of old nesting material, parasite load and internal illumination). To monitor hole-nesting bird populations properly, it is therefore important to determine if nestbox aging actually has an effect on nestbox selection and how significant the effect is, so that monitoring methods can be optimised. In Latvia, the practice of using nestboxes in ornithological studies has a long history. The first nest box surveys had already started by the late 1940s, and reached a peak of activity in the 1950s (Mihelsons 1958a, 1958b; Spuris *et al.* 1958, Mihelsons 1964). After an interruption of several decades, the studies were renewed in the 1980s in the form of a monitoring programme carried out in various parts of Latvia (Cauns 1990). By the late 1980s, there were *c*20 nestbox plots. Monitoring project data are used in the

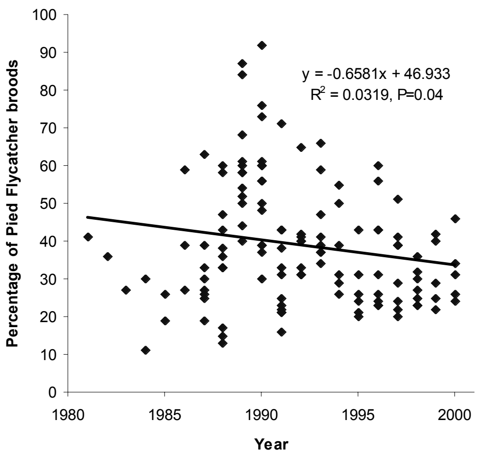


Fig. 1. Percent of nest-boxes occupied by Pied Flycatcher during the study period (1981-2000). Line indicates linear regression.

current study to analyse the impact of nestbox aging on the breeding of hole-nesting birds.

## 2. Study area and methods

The study plots were chosen in different parts of Latvia in pine forest stands on poor soils. There were 100 nestboxes in each study plot (except for 2 plots with 160 and 180 respectively). The nestboxes used were a special type designed by Maris Cauns to protect nesting birds from pine marten (*Martes martes*) (Vilka 1999). Nestboxes were placed generally at intervals of 35 m (some plots had 25 m or 50 m intervals). Amateurs as well as professional ornithologists collected data. The quality of data differs between study plots and years. Of all the nestbox plots (41) established during the monitoring project, only 20 have been used in the current data analyses (because the plots had regular intervals between nestboxes, the nestboxes were a standard type, and the plots had produced a sufficient level of data). Study plots were checked at least twice during the breeding season to obtain information on productivity (clutch size and fledgling numbers in each brood) and the number of broods of each species. During the survey period, the number of study plots monitored annually fluctuated between 5 and 14, and the number of nestboxes between 530 and 1400. In total, the data from *c*12 700 'nestbox seasons' (nestboxes × seasons) are used in the analyses.

Not every nestbox plot was established in the first year; the first being in 1981 and the last two in 1991. To estimate the effect of nestbox aging, the bird data were arranged by age after establishment of the corresponding study plots (1$^{st}$, 2$^{nd}$ year *et seq*). New nestboxes (100) replaced all old boxes in one study plot (Livberze) in 1996 (7 years after the plot had been established).

In the analysis of Pied Flycatcher productivity, broods with extreme numbers of eggs and nestlings (fewer than 5 or more than 8) were excluded from the study to avoid incomplete or repeat broods and to discount clutches made by more than one female. To estimate the population trends and the impact of nestbox age, simple regression analysis was used. To test each comparison (bird data versus years and bird data versus nestbox age) the significance level $\alpha/k$ (where k is number of comparisons; the Bonferroni method) was used.

## 3. Results

The breeding numbers both of Pied Flycatcher and Great Tit declined significantly in nestbox plots during the survey

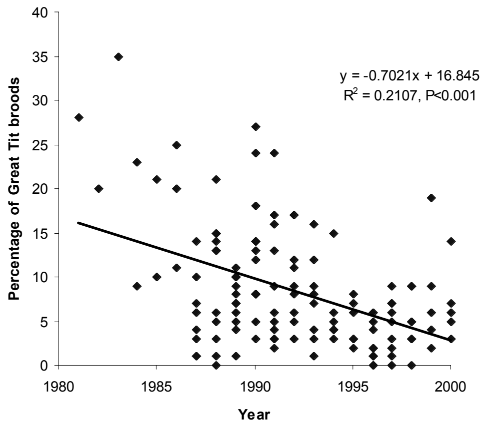


Fig. 2. Percent of nest-boxes occupied by Great Tit during the study period (1981-2000). Line indicates linear regression.

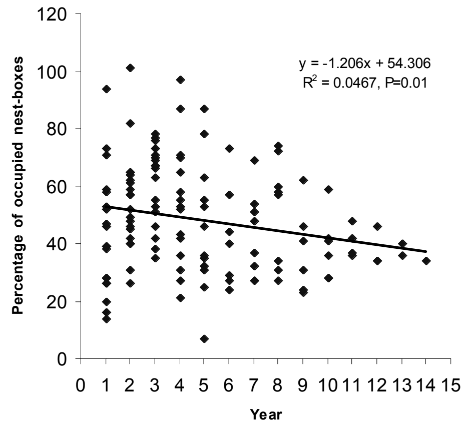


Fig. 3. Percent of occupied nest-boxes versus the age after the establishment of study plot. Line indicates linear regression.

period (Fig. 1 & 2). When the bird data were arranged by the corresponding age of nestbox plots, the percentage of nest boxes used by birds decreased significantly with nestbox age (Fig. 3). An increase in occupation of nestboxes was observed in the 2nd and 3rd year after the nestbox plots had been established, but a decrease then followed (Fig. 3). When the number of broods versus nestbox age were analysed separately for each species, there was still a negative trend for Great Tit (Fig. 5). The numbers of Pied Flycatcher also seemed to decrease with nestbox age (Fig. 4), but the decline was not significant.

If the factor of nestbox aging was taken into account in the estimation of population trends (using the Bonferroni method to test the comparisons), the negative trend in breeding numbers was still significant ($P<0.05/2$, Fig. 2) for Great Tit, but not for Pied Flycatcher ($P>0.05/2$, Fig. 1). The number of Pied Flycatcher broods increased after new nestboxes had replaced the old in the sole selected study plot in 1996 (1995, 26 broods; 1996, 43 broods). Over the same period, the number of Great Tit broods seemed to decrease (7 and 4 respectively). However, in 1996 the number of Pied Flycatcher broods also increased in other study plots: in 1995 the mean percentage of nestboxes occupied by Pied Flycatcher was 27.5 (6 study plots used) and in 1996 36.6% (9 study plots used).

No significant trends in the productivity of Pied Flycatcher were found during the survey period, but the clutch size was positively correlated with nestbox age (simple regression, $P=0.01$, R-squared=0.19, n=3134). The number of fledglings seemed to show a similar trend ($P=0.05$, R-squared=0.18, n=2060). No significant correlation was found between the productivity of Great Tit and nestbox age (simple regression, $p=0.4$ for eggs and $p=0.5$ for fledglings).

## 4. Discussion

Throughout most of Europe, breeding populations of Pied Flycatcher and Great Tit are considered to be stable, with small

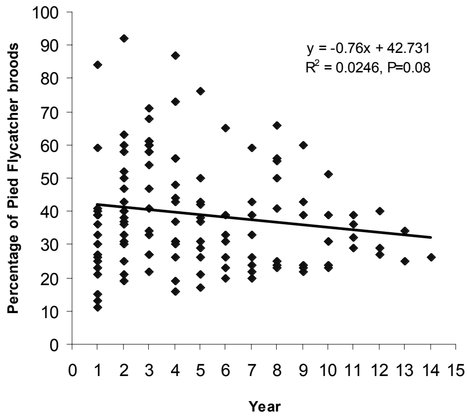


Fig. 4. Percent of nest-boxes occupied by Pied Flycatcher (% of all available nest boxes in the study plot in corresponding year) against the nest-box age (years after the establishment of study polt). Line indicates linear regression.

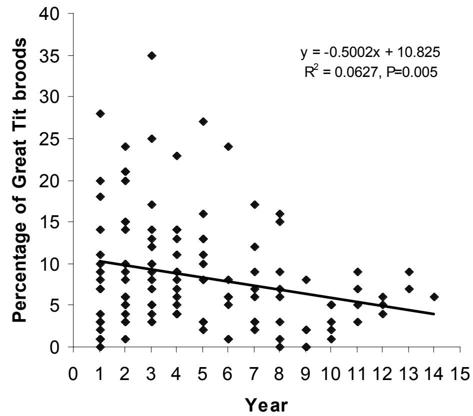


Fig. 5. Percent of nest-boxes occupied by Great Tit (% of all available nest boxes in the study plot in corresponding year) against the nest-box age (years after the establishment of study polt). Line indicates linear regression.

changes in numbers (Gosler & Wilson 1997, Lundberg 1997, Snow & Perrins 1998). Several authors have noted an increase in numbers of hole-nesting birds in the 1980s due to nestbox provision (Snow & Perrins 1998, Edula 1998). A decrease since the early 1980s in breeding numbers of both species has been observed in Latvia (Fig. 1 & 2). For the Pied Flycatcher, a similar trend has also been observed in breeding populations in Norway during 1986-1997 (Thingstad & Fjeldheim 1999) and in the numbers of birds migrating on the southern Baltic coast during 1961-1990 (Busse 1994). For the Great Tit a slight decrease has been registered in Estonia, but the population is still considered stable (Kuresoo & Ader 2000).

The probable decreases in numbers of Pied Flycatcher and Great Tit, as observed in the current study, could have been caused by a number of other factors. One such reason might be forest management activities in the study plot stands, for considerable area of 8 stands were thinned. However, both species are believed to exhibit a cyclic pattern of abundance (Busse 1994, Baumanis & Celmins 1993), and so a subsequent increase in numbers could follow. The number decrease may also have been influenced by peripheral side factors that are functions of the monitoring methods used (*e.g.* nestbox aging). Our study on the impact of nestbox age on the breeding of small hole-nesting birds suggests that nestbox age can affect brood numbers (Fig. 4 & 5).

Several European researchers had observed that hole-nesting birds exhibit a preference for new nestboxes over old (Shcherbakov 1956, Mihelsons 1958a, Blagosklonov 1970, Lundberg & Alatalo 1992). In a study of hole-nesting birds in central Russia, Blagosklonov (1970) observed maximum nestbox occupation in the 2nd year after establishment, followed by a gradual decrease. These results correspond with our current study (Fig. 3). In Latvia, the nestbox aging effect had already been observed in the 1950s by Mihelsons (1958a). Occupation of nestboxes made from fresh pine deal and sited in the Riga Forestry District in 1948, had already decreased by 1952. During spring 1954, in two forest areas where half of the nestboxes had been replaced by new ones, occupancy increased in the new boxes: in one district new occupancy rate was 65.5% and old 37.3%, in the other 73.3% and 48.3% respectively. When old and new nestboxes were sited in a new area, occupancy rate was 17.3% in the new and only 4.3% in the old (Mihelsons 1958a). Subsequently, one more study was carried out in another forestry district with similar results (Mihelsons 1958a). Mihelsons concluded that nest boxes made of fresh 25 mm-thick deal became less attractive for hole-nesting birds in the fourth or fifth year after establishment (Mihelsons 1958a).

Several reasons have been proposed to explain why hole-nesting birds prefer new nestboxes to those that are older but still usable (Mihelsons 1958a, Blagosklonov 1970, Lundberg & Alatalo 1992): presence of parasites, presence of old nesting material, microclimatic conditions, visibility (external brightness) and internal light conditions in the nestbox.

To investigate the role *parasites* play in prospecting birds avoiding old nestboxes, Blagosklonov (1970) removed more than 50 nestboxes, stored them dry for 2-3 years to kill off parasites, and then placed them again in the forest. Occupancy did not increase, suggesting that parasites were not the reason prospecting birds tended to avoid older nestboxes. Studies have also shown that the impact of *old nesting material* has no effect on birds' choice of nestbox. In several studies in Sweden, Pied Flycatcher show no preference for nestboxes lacking old nesting material (Alerstam 1985; as cited by Lundberg & Alatalo 1992), but did show a preference for new nestboxes containing old nest material to those without any nest material (Olsson *pers comm* as cited by Lundberg & Alatalo 1992). Changes of *microclimate* inside the nestboxes due, for example to development of cracks with age and accumulation of moisture, had also been suggested as a possible reason for avoidance of old nestboxes (Mihelsons 1958a, Lundberg & Alatalo 1992). However, in many cases, nestboxes that appear to be intact and quite suitable for breeding were not occupied by birds, while neighbouring nestboxes that had many cracks were (Mihelsons 1958a).

New nestboxes, being brighter, contrast with the background against which they are mounted and therefore birds may detect them more easily (Lundberg & Alatalo 1992). However, several researchers by their observations have denied the role of *external brightness* of nestboxes in nest-site choice. For example, in an experiment to catch Pied Flycatchers in spring when they are prospecting for nest sites, Mihelsons, using special nestboxes designed to capture birds (externally they resembled ordinary boxes), recorded similar numbers of birds in new and old boxes (Mihelsons 1958a). These results suggest that flycatchers can find old nest boxes as easily as new ones, and that *external brightness* could not be the main reason for them avoiding old nestboxes as nesting sites. To check the role of *internal light conditions* in nestboxes, Blagosklonov (1970) blackened the inside of nestboxes, without changing the external colour. Pied

Flycatchers showed an obvious preference for boxes with a bright interior (Blagosklonov 1970). Importance of *internal light conditions* in nestboxes in association with the depth of the nestbox had been observed earlier by Shcherbakov (1956) in the Mordovian Nature Reserve. According to his observations, Pied Flycatchers prefer shallower nestboxes than Great Tits. When Pied Flycatchers found nestboxes to be too deep, they filled them with nest material to the level they preferred (Shcherbakov 1956). Based on Shcherbakov's (1956) findings and on his own measurements and calculations, Mihelsons suggested that the internal reflectance of the nestbox is of great importance to the Pied Flycatcher in selecting a nestbox in which to nest. Pied Flycatcher avoided selecting those nestboxes in which daylight could illuminate the nest directly and those whose internal reflectance was too low (nestboxes made from dark materials). However, to the Great Tit, a nestbox that had low interior reflectance did not affect greatly its selection as a nest site (Mihelsons 1958a).

Blagosklonov (1970) showed that Great Tit preferred deep nestboxes if they were new, but shallower boxes as these became older. He concluded that the level of *internal light conditions* was the most important factor for hole-nesting birds in their selection of nestboxes as nest sites. Blagosklonov (1970) also noted the part that entrance hole and depth played in determining the light levels inside the nestbox.

He noted that internal light levels diminish as the nestboxes aged, the amount of light that the internal walls reflected reducing as the brightness reduce, thus placing the nest in increasing darkness from season to season. Blagosklonov (1970) confirmed this idea by making measurements of light intensity. He tested this further by dividing into pairs those nestboxes that had been in place in the forest for 7-8 years and painting white the rear inside wall of one of each pair. He put up a third set of nestboxes newly made from fresh wood. The old nestboxes having a white inside rear wall all became occupied, but occupancy of the unchanged old boxes was only 37% and of the new boxes, 71%. In another experiment, he blackened the internal walls of 20 new nestboxes, the other 20 new boxes serving as controls. Pied Flycatcher avoided selecting blackened boxes for nesting (only 5% were occupied), preferring the control boxes (80% occupancy) (Blagosklonov 1970).

According to the above literature it would seem that reduced *internal lighting conditions* could be the main reason for the nestbox aging effect. The current study suggests that the aging of nestboxes affects Great Tit more than the Pied Flycatcher. In the study by Mihelsons (1958a), Great Tit also seemed to prefer new nestboxes to old ones. Possibly, the *internal lighting conditions* are not the sole factor causing the aging effect of nestboxes on nest site selection. It should be noted that the significant negative trend of Great Tit breeding numbers in the nestbox plots during the study period presented difficulties in estimating the impact of nestbox aging on nest site selection.

In the current study, after nestbox replacement in one of the study plots, the number of Pied Flycatcher broods increased, but the number of Great Tit broods did not, circumstances that support suggestions that Great Tit prefers deeper nestboxes that characteristically have poorer *internal lighting conditions*, unlike the Pied Flycatcher (Shcherbakov 1956). Although the number of Pied Flycatcher broods in 1996 also increased in other study plots compared to 1995 levels, the increase in the study plot where the nest-boxes had been changed was even greater.

It may seem surprising that Pied Flycatcher productivity increases with nestbox age. One explanation could be the occupancy of new nestbox plots by young (2nd summer) first-time breeders, which had, after fledging the previous year, dispersed widely and have no strong connection to their natal area (Mihelsons, 1958b). Inexperienced breeders have a lower productivity than older birds (Haartman 1951, Berndt & Winkel 1967). Another explanation could be the density dependence of productivity (Stenning *et al.* 1988, and Virolainen 1984, both as cited by Newton 1998). Occupancy of nestboxes in new study plots was higher (Fig 3) than in older plots, leading to higher breeding density and hence reduced productivity (Stenning *et al.* 1988,and Virolainen 1984, both as cited by Newton 1998).

Results of the analyses of our hole-nesting bird monitoring data and the literature data show that nestbox aging seems to be an important factor influencing nest site selection and, by definition, monitoring results. Our present knowledge allows us to recommend that nestboxes should be replaced by new ones every 3 to 4 years. However, the best replacement schedule should be established to minimise the impact of nestbox aging on the results of any monitoring scheme. Such a schedule should be derived from the results from study plots containing nestboxes. More research into the importance of *internal light conditions* that a bird species requires in a nestbox to select it as a nest site is needed. Such research would require techniques developed to minimize the reduction of *internal light conditions* that arises as the nestbox ages.

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# Breeding Bird Monitoring in France: The ACT Survey

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## 1. Introduction

The ACT (Alaudidae, Columbidae, Turdidae) monitoring plan is a project of the French National Game and Wildlife Agency (ONCFS, Office National de la Chasse et de la Faune sauvage). Its main objective is to monitor the breeding populations of twelve migratory bird species in France (Boutin *et al.* 2001). These species are: Blackbird *Turdus merula*, Mistle Thrush *Turdus viscivorus*, Fieldfare *Turdus pilaris*, Song Thrush *Turdus philomelos*, Ring Ouzel *Turdus torquatus*, Woodpigeon *Columba palumbus*, Stock Dove *Columba oenas*, Turtle Dove *Streptopelia turtur*, Collared Dove *Streptopelia decaocto*, Skylark *Alauda arvensis*, Woodlark *Lullula arborea* and Quail *Coturnix coturnix*.

## 2. Method

From an experimental feasibility study that was carried out in 1992 and 1993, a protocol, based on point counts (Blondel *et al.* 1970, Frochot & Roché 1990) was selected and applied at a national scale. Every year, there were 1000 counting routes, each including 5 listening points. No point count could lie closer than 1000 m from any other. Originally, the count duration was 20 minutes (1993 to 1996), but from 1996 onward it was 10 minutes. The shorter duration of 10 minutes was preferred because it was a more efficient use of manpower. 800 observers covered these routes in a network called 'Birds of passage' in collaboration with hunting association teams. These observers are not amateurs but professionals from our technical staff. Each observer makes two visits each year, the first being census between 1st to 30th April and the second between 15th May and 15th June. The basic units of count are singing males, but birds detected by flight are also recorded. It is therefore a simple bird count with no estimation of distance, because the aim of the ACT monitoring plan is to provide an index of relative abundance for each species.

## 3. Results

The data obtained had been gathered in the first seven years of the ACT plan, from

Tab. 1: Records of the most common species in 2000.

| Species | Points | Singing males | Average |
|---|---|---|---|
| Blackbird | 4505 | 8312 | 1.845 |
| Woodpigeon | 4505 | 4702 | 1.044 |
| Skylark | 4505 | 3765 | 0.836 |
| Song Thrush | 4505 | 2190 | 0.486 |
| Collared Dove | 4505 | 2348 | 0.521 |
| Turtle Dove | 4505 | 2896 | 0.421 |
| Mistle Thrush | 4505 | 1214 | 0.269 |
| Quail | 4505 | 304 | 0.067 |

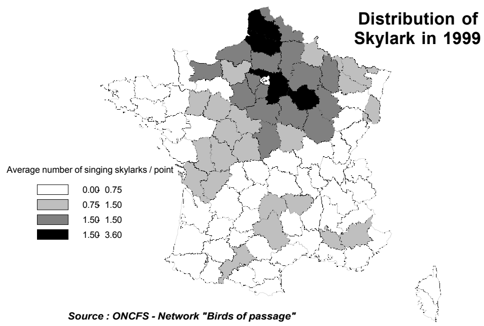


Fig. 1. Distribution and abundance of the Skylark Alauda arvensis in spring 1999.

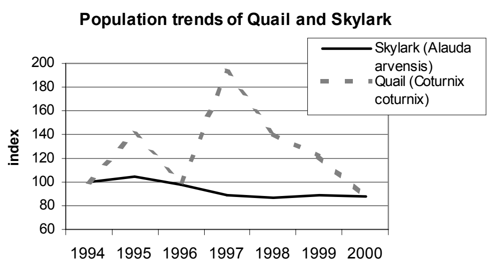


Fig. 2. Trends in Quail and Skylark in France.

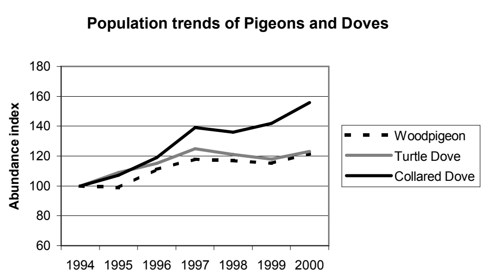


Fig. 3. Trends in Columbidae in France.

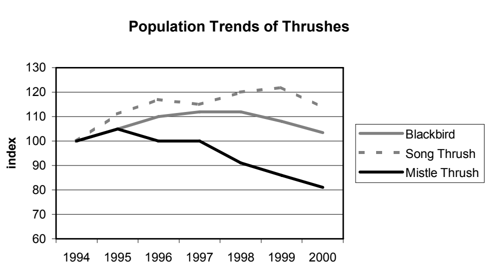


Fig. 4. Trends in Turdidae in France.

1994 to 2000. The average number of singing birds per point is considered to be the best indicator of abundance during the breeding period. These data, analysed at the *departemental* or regional scale, reveal the areas in France that are important to these species.

The data in Figs 3-4 provide the population trends for eight common species in the 1994 to 2000 study period, on the basis of a theoretical index of 100 in 1994. For the other species the field data are too scarce to allow powerful analyses.

## 4. Conclusions

Two species are decreasing; the Skylark whose national abundance index has gone down from 100 to 88, and the Mistle Thrush. The Columbidae trends are increasing (Boutin 1998) and above all, that of the Collared Dove. The Blackbird and the Song Thrush have shown an increase early in the monitoring period and a subsequent period of stability or slight decrease. The fluctuations of Quail are pronounced. In a second step, the data will be linked to habitat types for an in-depth study of the reasons and to provide propositions for management actions (Eraud & Boutin 2001). Because the covariables are used for the geographical scales and habitat types, we will use Trim software (ter Braak *et al.* 1992, Pannekoek & van Strien 1998) to analyse the data.

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# Ornithological databases for science and conservation - management and project oriented studies

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Since 1992 the ornithological working group of Upper Austria has maintained an ornithological database at the biology centre of the Museum of Upper Austria. Data comprising 375 000 records about bird distribution are now included in and managed by the Museum's ZOBODAT professional database-system (Malicky & Aubrecht, 2002). Applying standardised questions yields concentrated information about taxonomic, geographic and conservation oriented contents (www.biologiezentrum.at).

Two examples demonstrate the system's capability:

1) For the project 'Upper Austrian Breeding Bird Atlas 1997-2001', we required rapid annual documentation of areas needing further investigation. Imputed data are analysed to project the number of species per grid unit. The resulting numbers are ranked by applying the species-area relationship as a working hypothesis. The resulting grid-map presents low and zero numbers that show areas needing further investigation as a priority. The whole process is computerised and does not require any additional manual procedures.

2) In the development of the project 'Threatened Meadow Bird Conservation', a scheme that began in 1992, a database query produced a map showing the cumulative distribution of *Crex crex, Saxicola rubetra, Anthus pratensis, Gallinago gallinago* and *Numenius arquata* in Upper Austria. The project has identified areas of dense distribution and has initiated much fieldwork (*e.g.* co-operation of regional working groups and the involvement of BirdLife International and WWF) from which a site-based monitoring programme was derived (Uhl 1993). The results were used to evaluate IBAs in Upper Austria. 'Maltsch', one of the most important areas situated at the Czech border, was declared as NATURA 2000 site. WWF and the Czech authorities develop and implement bilateral management plans, assisted by incorporating the Interreg II project GREVOLATO (Uhl *et al*. 2000, Uhl 2001).

In both projects analyses of bird numbers from survey and monitoring studies are used to quickly identify gaps in knowledge or to verify and monitor sites of conservation concern.

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# Pan-European monitoring of Important Bird Areas

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## 1. The BirdLife International Important Bird Area (IBA) Programme

The Important Bird Areas (IBA) Programme of BirdLife International is a worldwide initiative aimed at identifying and protecting a network of core areas (or 'critical sites') for the conservation of the world's birds. The first IBA inventory to cover the whole of Europe was published in 1989 (Grimmett & Jones 1989). Facilitated since 1990 by a coordinator at the BirdLife International Secretariat and, increasingly, by national IBA Coordinators in individual countries, the actions of many individuals and organisations have coalesced into a large-scale European IBA Programme. This has resulted in the production of twenty national IBA inventories, and more recently, a new pan-European inventory (Heath & Evans 2000). As at October 2001, a total of 4000 IBAs had been identified in Europe. Building on this inventory of sites, the European IBA Programme for the last decade has addressed site-oriented research and action, encompassing habitat management, education, advocacy, national and international legal protection, and monitoring, the focus of this article.

## 2. Monitoring IBAs

IBA monitoring is one of three themes central to the Pan-European Bird Monitoring

Tab. 1. Core indicators used for measuring the conservation status of IBAs in Europe and data availability.

| Indicator type | Indicator | Description | N$^o$ records in WBDB* |
|---|---|---|---|
| State | Site boundary and area | Paper or digital map of each IBA boundary and a measure of its area (hectares). | 4000 |
| | Habitat | Inventory of all primary habitats (10 types) covering >5% of each IBA and the total area of each type within each IBA. | 12 000 |
| | Key bird populations | Population size and trend (during past 10 years) of each bird species for which each IBA was selected (average of 4 species per IBA). | 17 000 |
| | Land-use | Inventory of all land-use (12 types) covering >5% of each IBA and the % cover of each land-use within each IBA. | 12 000 |
| Pressure | Threats | Inventory of key threats (12 types) and their impact (using standard IBA methodology) within each IBA | 14 000 |
| Response | Protection status | Inventory of over-lapping protected areas and the extent of over-lap between each IBA and the protected area | 16 000 |
| | Management plan | Whether each IBA is (partly or wholly) covered by an existing management plan | 650 |

*WBDB - BirdLife International World Bird Database (a purpose-built database used for the management, analysis and reporting of data held by BirdLife International).

Tab. 2. Draft monitoring and reporting cycle for IBA core indicators.

| Indicator type | Indicator | Proposed monitoring and reporting cycle | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | 2002 | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 |
| State | Site boundary and area | x | x | x | x | x | x | x | x | x | x | x |
| | Habitat | | | x | | | | x | | | | x |
| | Key bird populations | x | x | x | x | x | x | x | x | x | x | x |
| | Land-use | | | x | | | | x | | | | x |
| Pressure | Threats | x | x | x | x | x | x | x | x | x | x | x |
| Response | Protection status | x | x | x | x | x | x | x | x | x | x | x |
| | Management plan | x | x | x | x | x | x | x | x | x | x | x |

Strategy. Monitoring is necessary to track the state of IBAs, the pressures acting upon them and the responses being made to conserve them. The results of monitoring are needed to inform decisions, especially those concerned with to setting conservation priorities, diagnosing problems and predicting change. There are several benefits to be derived from coordinating IBA monitoring and having an agreed set of common standards at a European scale:

- At a local level, staff have a framework within which they can develop their programme of site monitoring and can have confidence that it is supported and being implemented at a European level.
- If data are collected, managed and exchanged following accepted standards, the costs of data exchange are substantially reduced. Less time is spent interpreting data from different sources, allowing prompt comparison of results.
- Common standards allow data to be provided at the right time and in the right format. The data can then be aggregated and information produced at a range of geographical scales. This enables reporting on the conservation status of IBAs at regional, national and continental levels.

Standards for monitoring core areas need to be sufficiently robust so that they can be implemented consistently across Europe by different organisations, yet flexible enough to cater for the different operational practices and systems that have evolved in each country. There are two main elements of common standards for monitoring IBAs:

- Indicators.
- Monitoring cycle and reporting.

## 3. Indicators

The conservation status of an IBA is complex and comprises many interacting variables. Hence, any monitoring scheme needs to identify which data are robust enough to allow simplification into a set of core data that are easily measured, understood and communicated, and are scientifically sound. For this purpose, indicators are used. Indicators must signal key issues to be addressed through interventions and other actions, and hence build a bridge between the fields of policy and science. Once selected, they give direction to monitoring and research programmes.

In order to be effective, an indicator must:

- Quantify information so that its significance is apparent.
- Be user-driven (to help summarise

information of interest to the intended audience).
- Be scientifically credible.
- Be responsive to changes in time and space, or both.
- Be simple and easily understood by the target audience.
- Be based on information that can be collected realistically.

Indicators are generally developed within a framework that incorporates three main categories:
- State: quantity and quality of IBAs (*e.g.* population levels of key bird species).
- Pressure: threats to IBAs (*e.g.* impact of hunting).
- Response: conservation of IBAs (*e.g.* % of area under legal protection).

The conservation status of IBAs has been measured using seven core indicators, which have generated a large amount of data to date (Tab. 1). Most of these data are available online at http://www.birdlife.org.uk/sites.

## 4. Monitoring cycle and reporting

There is a need to monitor and report on indicators within an agreed cycle. This cycle needs to take account of the scale of monitoring required, the likely rate of change in indicator levels and reporting needs. For IBAs, the most obvious cycle to operate within is the 4-year cycle of the BirdLife International Global Partnership Meetings, for which the next meeting is scheduled for 2004. The draft cycle being discussed presently within the BirdLife Partnership would ensure that five of the seven core indicators would be re-measured annually in all IBAs and at least once every four years for the remaining two indicators (Tab. 2).

Reports on the state and trends of indicators at IBAs are required for a variety of purposes and on a variety of scales. The common standard is to allow the separate country accounts to be compared and aggregated to produce a European account on the overall state of IBAs, the pressures acting upon them and the action being taken to conserve them. This would feed into a new indicator-based report on the state of the world's birds and IBAs that will be launched at the next BirdLife Global Partnership Meeting in 2004.

## 5. Problem analysis

At a 1-day workshop held in Gibraltar in September 2001, IBA Coordinators from most European countries analysed the problems facing the achievement of the monitoring and reporting cycle\ proposed in Tab. 2. Problems of course varied greatly between countries, but included often inadequacies in:
- Practical methods for measuring indicators.
- Numbers of skilled field workers.
- Clear reporting procedures.
- Design and use of data management systems.
- Inter-organisational coordination of monitoring schemes.

## 6. Conclusions

Currently, BirdLife International is working (together with others) to overcome the problems noted above so that IBAs can be monitored on a pan-European scale. With the use of the existing systematic information base, which will be strengthened through monitoring, the ultimate prospects for greater, more complete and more durable success in the conservation and wise use of IBAs in Europe are better than ever before. The challenge for governments and NGOs in a position to put this data to use is to make these prospects reality. BirdLife urges all executive agencies to apply the highest levels of commitment to this aim, and stands ready to collaborate wherever and whenever possible.

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# The Breeding Bird Atlas of Jelgava district, Latvia

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## 1. Introduction and Methods

The district of Jelgava lies in the middle of Latvia and has a total area of 1613 km$^2$. Forests cover ~30 % of the territory, bogs ~5% and agricultural lands ~60%. Mapping the bird distribution was by traditional atlas-type methods (Priednieks *et al.* 1989). The territory was investigated using 5 km×5 km UTM squares (n=93). The evidence of a species breeding in a square was estimated according to 17 categories divided into 4 grades representing presence of the species, possible breeding, probable breeding and confirmed breeding. 91 squares (97.9%) were surveyed during the 1995 to 1998 breeding seasons. 67 squares were covered sufficiently (at least 15 of the 20 commonest species were detected). Two national border squares whose total area was small (<5km$^2$) were not visited.

## 2. Results

A total of 162 bird species was registered, out of which 114 species are confirmed breeders, 31 – probable breeders and 17 possible breeders. 9 species were recorded while only feeding or on passage migration and were omitted from the list of nesting birds. The largest number of nesting species registered in a single square was 107. Breeding population numbers were estimated and additional data were collected on rare and colonial species. Data for some species were compared to the data from the 1980-84 countrywide atlas (Priednieks *et al.* 1989). Changes in numbers are mostly usually connected with the recent crash of intensive agriculture (after the end of the Soviet occupation in Latvia) that produced large areas of undisturbed semi-natural landscapes.

**Species of interest**

White Stork *Ciconia ciconia*. It is a very common species. The calculated number of occupied nests in the Jelgava district nest surveys of 1994-1995 was 415, the average nest density being 43/100 km$^2$. Overall numbers in Latvia have also increased considerably (Janaus & Stípniece 2000).

Mute Swan *Cygnus olor*. The increase in numbers is caused by its recent (since the early 1980s) breeding range expansion. Before then there had been no Jelgava breeding records. Recently, 57% of approximately 25 bp (breeding pairs) nested beside water-filled open clay pits, 23% by artificial reservoirs or channels, and the rest in natural habitats along the Rivers Lielupe and Svēte.

Marsh Harrier *Circus aeruginosus*. Numbers have increased since the early 1980s and although clear reasons are not known, the increased area of abandoned agricultural lands and encroachment of disused open clay pits might be factors.

Montagu's Harrier *Circus pygargus*. The considerable increase in numbers is most likely caused by the abandonment and the enforced low-intensity use of agricultural lands in the 1990s. Breeding sites include open, unused and overgrown clay pits and wet meadows along the rivers.

Lesser Spotted Eagle *Aquila pomarina*. Numbers of breeding pairs might have increased in the early 1990s, but forest management practices in small private forests have caused destruction of nests and habitat in the late 1990s. The total number of breeding pairs in Jelgava district (Fig. 1) might reach 20 (2.8-3.8 bp/100 km$^2$ of forest (Strazds *et al.* 1997)).

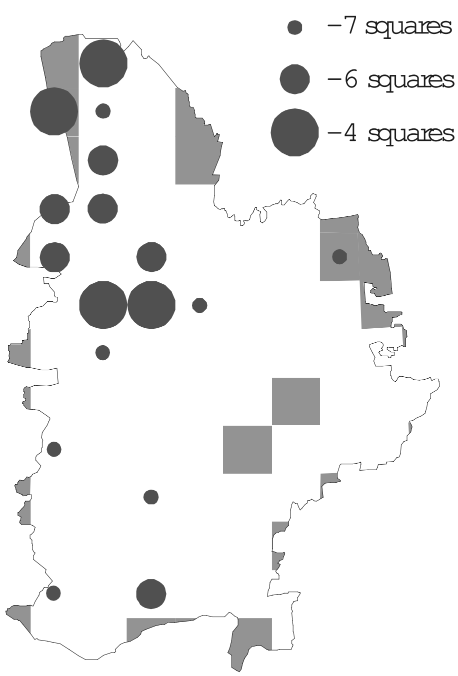


Fig. 1. Distribution of Lesser Spotted Eagle (*Aquila pomarina*) in the district of Jelgava 1995-1998.

Common Kestrel *Falco tinnunculus*. The species does not seem to have benefited from the crisis in agriculture and appears to be still decreasing in numbers. Locally fairly common, some counties having 3-4 breeding pairs, a level that is low in comparison with the mid 20$^{th}$ century (Vilks 1986).

Corncrake *Crex crex*. Numbers have increased in comparison with the early 1980s, but precise estimates of Jelgava district numbers are as yet unavailable. Floodplain meadows of the rivers Lielupe, Svēte and Iecava hold some of the highest breeding densities in Latvia, 18.75 calling males/km$^2$ (Keišs 1996).

Citrine Wagtail *Motacilla citreola*. Since this species was first observed in

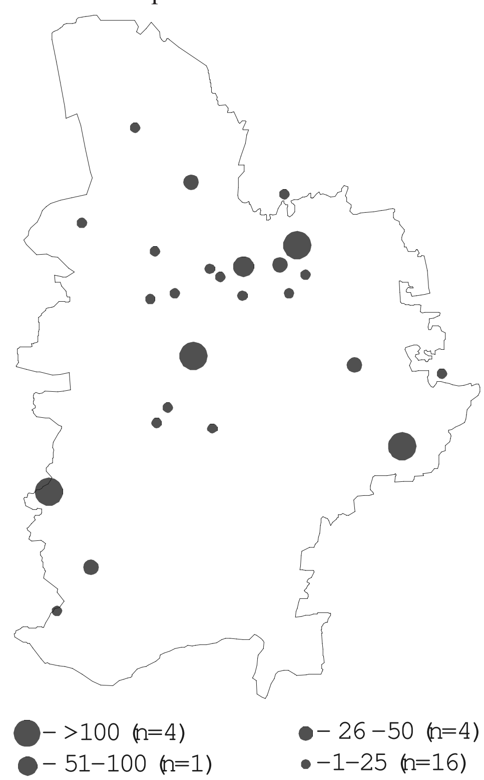


Fig. 2. Size of colonies of Sand Martin (*Riparia riparia*).

Tab. 1. Population estimates (1995-1998) and trends (1980-1984 v 1995-1998) of some bird species of the district of Jelgava, Latvia.

| English name | Species name | Population estimate (pairs) | Trend |
|---|---|---|---|
| Eurasian Bittern | *Botaurus stellaris* | 5–6 | 0 |
| Black Stork | *Ciconia nigra* | 10–15 | 0 |
| White Stork | *Ciconia ciconia* | >400 | +2 |
| Mute Swan | *Cygnus olor* | 20–25 | +2 |
| Marsh Harrier | *Circus aeruginosus* | 30–50 | +1 |
| Montagu's Harrier | *Circus pygargus* | 5–10 | +2 |
| Lesser Spotted Eagle | *Aquila pomarina* | 10–20 | +1 |
| Common Kestrel | *Falco tinnunculus* | 10–15 | –1 |
| Common Crane | *Grus grus* | 7–10 | +1 |
| Black-tailed Godwit | *Limosa limosa* | >15 | 0 |
| Redshank | *Tringa totanus* | ~50 | 0 |
| Common Gull | *Larus canus* | 5–10 | 0 |
| Black Tern | *Chlidonias niger* | 30–50 | –1 |
| Kingfisher | *Alcedo atthis* | >20 | 0 |
| Sand Martin | *Riparia riparia* | >1000 | –1 |
| Citrine Wagtail | *Motacilla citreola* | >5 | +2 |
| Penduline Tit | *Remiz pendulinus* | 70–100 | +2 |

Trends are summarized in Column 4, where +2 = an increase of more than 50% from the total population size in 1980–1984, +1 = an increase of 25-50%, 0 = stable population and –1 = a decrease of 25–50%.

Latvia in1982 in Jelgava district (Bergmanis 1984), it has expanded its breeding range and has bred regularly at several Jelgava sites since the early 1990s. First breeding was recorded in 1989 (E. Račinskis *pers comm*) and the number of breeding pairs has increased since then.

Sand Martin *Riparia riparia*. During the study period, 25 colonies of this species were found (Fig. 2), The majority being in open clay pits. An increase of vegetation succession on the pit banks might have caused a slight decrease in the numbers.

Penduline Tit *Remiz pendulinus*. Numbers of breeding pairs have increased considerably since the 1980s. The species appears to be continuing its breeding range expansion (Lipsbergs 1971) in Latvia, populating new sites that it has not used previously.

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# Four decades of waterfowl counts at pre-alpine Lake Constance

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## 1. Introduction

Lake Constance, a pre-alpine lake of 571 km$^2$ bordering on Austria, Germany and Switzerland, is one of the most important breeding, staging and wintering sites for waterfowl in Central Europe (Fig. 1). Coordinated counts were initiated as early as 1951, but since 1962 counts of all waterbirds at the lake take place once monthly from September through April (coordinated by the international Ornithological working group OAB). The present paper summarizes some of the results obtained so far.

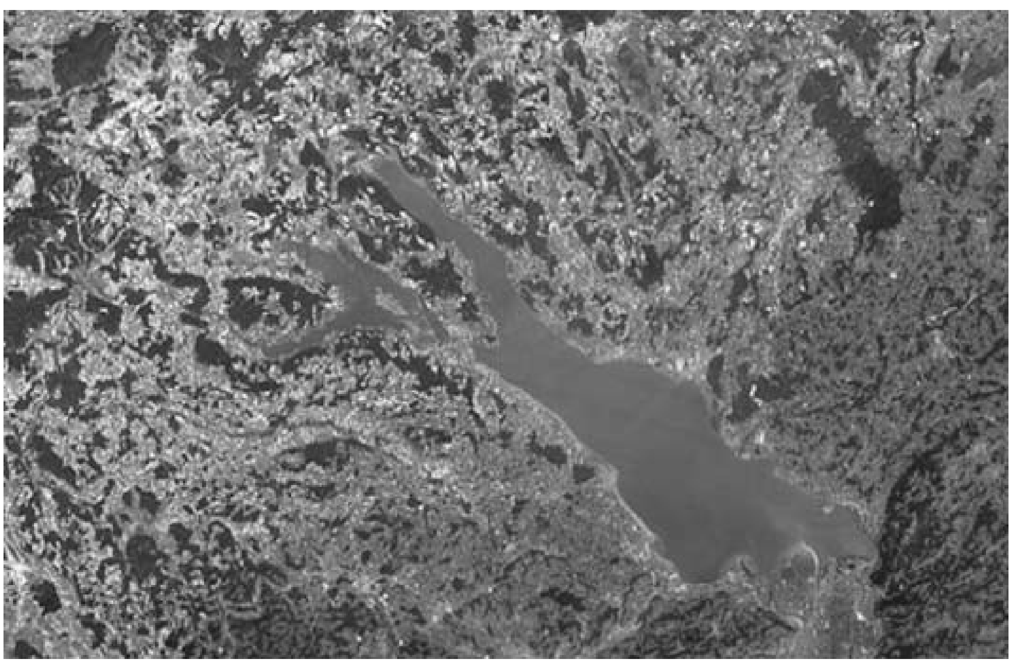

Fig. 1. Lake Constance lies at 400 m asl and features two clearly different ecological entities: a shallow, nutrient-rich western part (Untersee) covering 71.5 km$^2$, and a large, rather deep (max depth 254 m) eastern part (Obersee) covering 500 km$^2$ which is poorer in nutrients. The lake's water regime is largely dependent on the alpine system of the river Rhine that enters the lake at the Austrian-Swiss border..

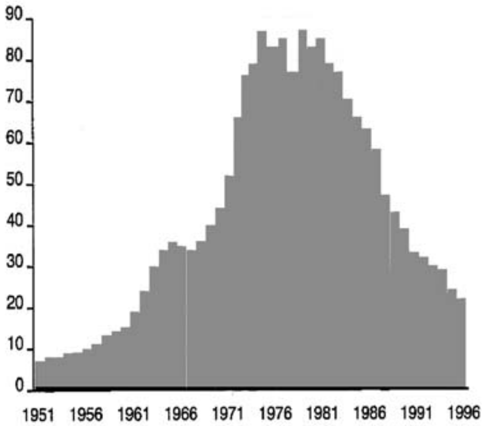


Fig. 2. Development of phosphorus concentrations (total amount of P) from 1951-96 at Lake Constance. The diagram shows that nutrient content in the lake increased dramatically in the 1970s to cause a considerable impact on lake ecology, especially on submerged macrophytes such as stoneworts (Characeae). From the 1980s onwards, nutrient levels in the lake were reduced markedly, allowing Characeae to recover.

## 2. Material and Methods

Volunteers carry out the counts from given points in 96 fixed segments around the lake. The counting always takes place on

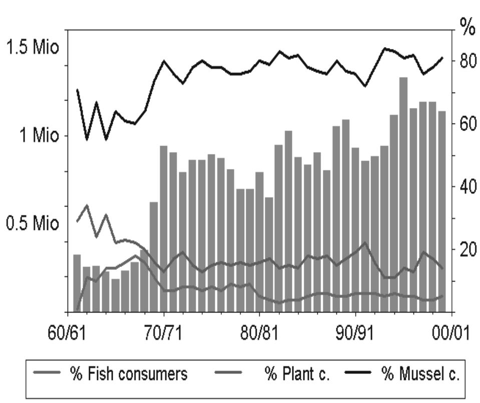


Fig. 3. Winter totals (September-March) of all wintering waterbirds 1961/62 through 1998/99.

the Sunday closest to the 15th of the month from September to April. The database available now comprises some 25 000 "segment counts". Analysis took place using standard software. Limnological data were made available by the Limnological Institutes of Langenargen and the University of Constance. Data on lake morphology were taken from the Geographical Information System BUGIS of the University of Hohenheim; other sources for the data analysis are described in Stark *et al.* (1999).

## 3. Results

Lake Constance's international importance may be assessed from two factors:

a. The maximum number of waterbirds recorded per month, annual numbers ranging to some 250 000 birds in October, and over 200 000 from November to January.
b. The 'winter totals' (*i.e.* all waterbirds counted from September to April), over a million birds in recent years (maximum: 1.3 m in 1995/96; see Fig. 3).

The dominant species are Pochard *Aythya ferina*, Tufted Duck *A. fuligula* and Coot *Fulica atra*, which comprised some 60% of all birds in winter in the 1960s, and up to 80% at present. Fig. 4 shows that Coot was the most dominant 'diving' species in the early years, but that Tufted Duck has become the most numerous since the 1980s.

The dramatic, fourfold increase in numbers of wintering waterfowl can be explained by two effects from the 1960s: the invasion of the lake by the zebra mussel *Dreissena polymorpha* and the massive degree of eutrophication (mainly by phos-

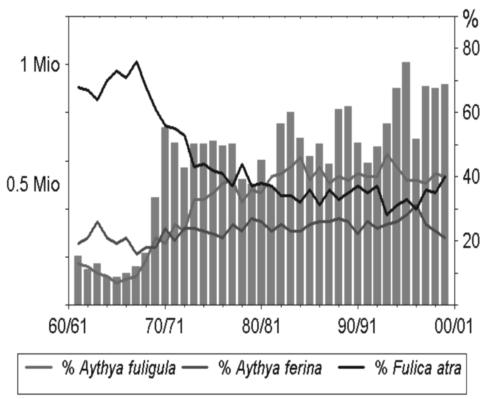


Fig. 4. Population dynamics of Tufted Duck *Aythya fuligula*, Pochard *A. ferina* and Coot *Fulica atra* in relation to the winter totals of all waterbirds consuming zebra mussels Dreissena polymorpha 1961/62 - 1998/99.

phorus, see Fig. 2). Mussel exploiters profited most from these changes in lake ecology, but fish eaters such as Great Crested Grebe *Podiceps cristatus* also increased considerably. The correlations between white fish (*Cyprinid spp*) catches and Great Crested Grebe numbers is highly significant (Bauer *et al.* 2000). One of the few species adversely hit by eutrophication was Red-crested Pochard *Netta*

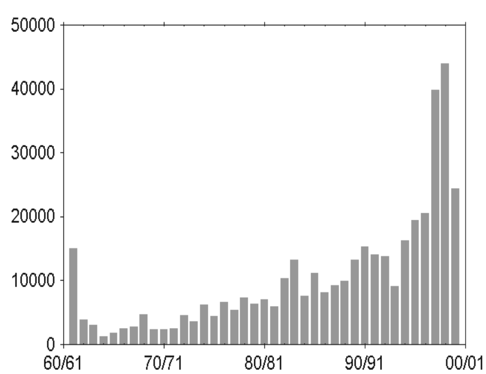


Fig. 5. Winter totals of Red-crested Pochard Netta rufina at Lake Constance from 1961/62 - 1999/2000.

*rufina* due to its dependence on submerged macrophytes (*Chara spp*) in oligotrophic conditions. Intensive water purification measures since the 1970s, reflected in a decrease of phosphorus (Fig. 2), have led to the re-emergence of large *Chara* banks, and Red-crested Pochard numbers have reached an all-time high in the last few years (over 20 000 individuals in Oct 99), winter totals reaching 40 000 individuals (Fig. 5).

## 4. Discussion

Four decades of waterbird counts by volunteers at a large and internationally important site offer unique opportunities for analyses. This papers sums up some of the results of a first preliminary analysis. It is shown that waterbird numbers reflect some of the dramatic ecological changes that took place at the lake during the last 40 years, but that the latest change, re-oligotrophication, has still to act on mussel consumers. Our intention is that a closer inspection of the data and a more elaborate use of limnological parameters will make fuller use of this enormous dataset (suggestions are very welcome). In a joint project with both Limnological Institutes at the lake, the importance of waterbirds in this lake ecosystem will be more intensively studied in future (Bauer & Stark 1999). Furthermore, the waterfowl counts at Lake Constance will continue unabated.

*Acknowledgements*. We are extremely grateful to the hundreds of volunteers and to the six coordinators who contributed to this unique data collection over these four decades.

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# 15 years of monitoring of 'large' gull species in central and northern Moravia, Czech Republic

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## 1. Study areas and methods

The study of large gull occurrence took place in two areas (Fig. 1). The first is the relatively natural landscape of central Moravia (left-hand rectangle) in the upper Morava river valley, which is dominated by fields and meadows. Sites indicated of particular interest are the Šumvald fish-pond (1), the Chomoutov gravel-pit (2), the Tovaeov fishponds (3) and the Záhlinice fish-ponds (4). The second area is the industrial area of the Ostrava-Karvina region (right-hand rectangle), which contains mainly artificial habitats typical of a coal-mining landscape. The primary counting sites were at the regular gull roosts at Doubrava industrial lake

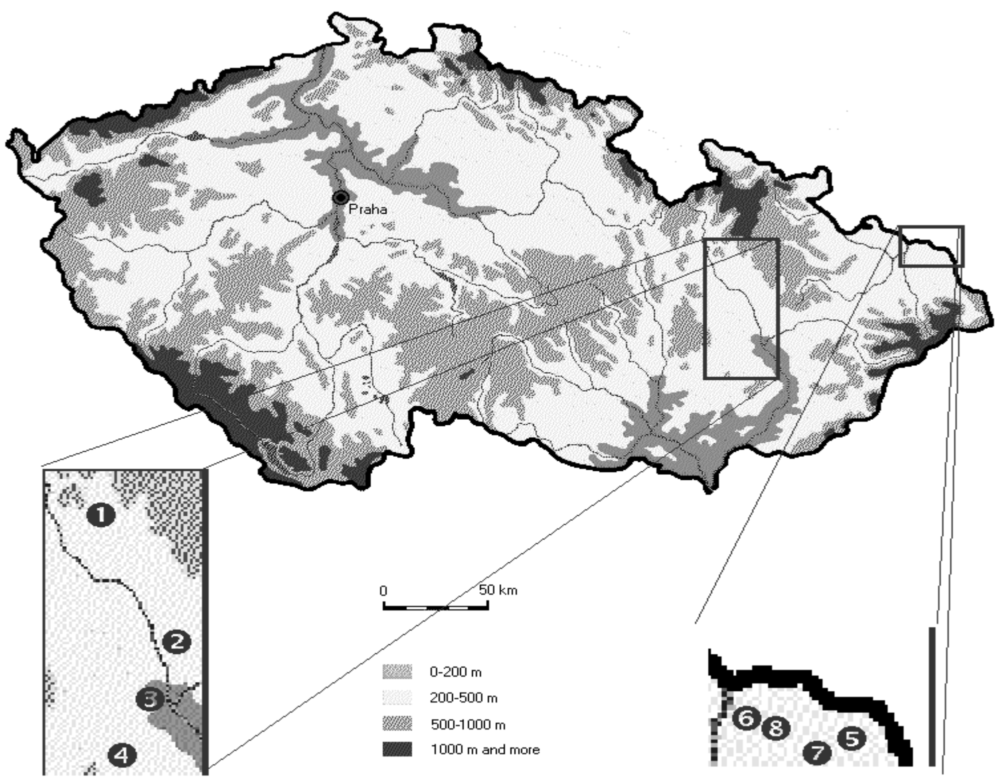


Fig 1. Study areas in the Czech Republic. See text for explanations.

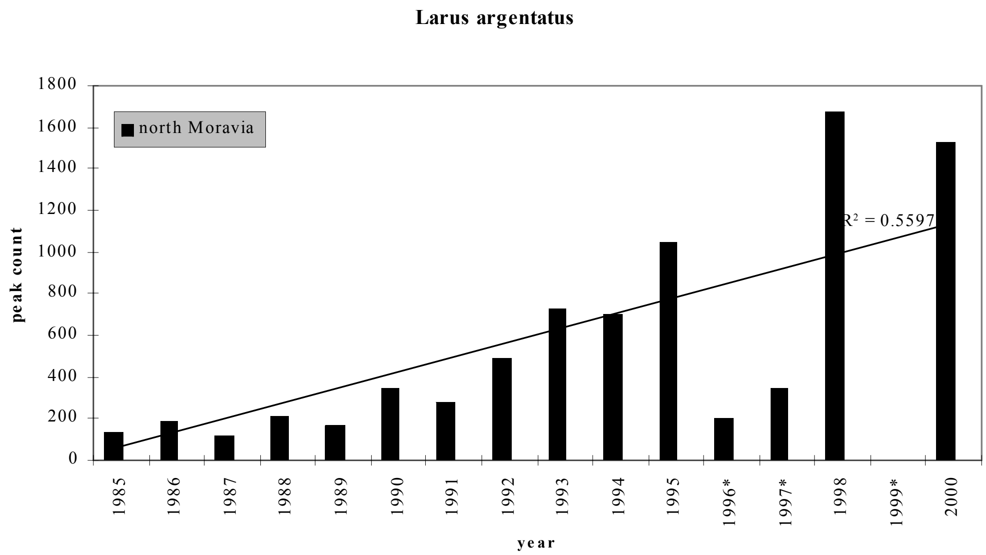


Fig. 2. Population size of *Larus argentatus* between 1985-2000.

(5), the Ostrava refuse tip (6), the complex of industrial lakes within which lie the Karvina and Havíoov refuse tips (7) and the Heomanice fishpond (8). Both study areas were checked regularly by myself and by several birders who have cooperated with me since 1985. The degree of co-operation was further improved in 1992 by the establishment of the Group for the Research of Larids (GRL). I collected the count results and published them regularly in the Bulletin of the GRL.

## 2. Results

### Herring Gull *Larus argentatus*

There is very large difference in the results obtained from the two study areas. In central Moravia, no significant change in numbers of Herring Gull was recorded. This species remains very uncommon in this area, only single birds or small flocks occurring there. In northern Moravia, Herring Gull has occurred in some numbers since the 1970s, the maximum being *c*100 birds. During the study period, a significant increase was recorded (Fig. 2). In the early years of the study, up to 210 birds visited a roost at Karviná, just a few hundred metres from the local refuse tip. Subsequently, numbers increased to about 700 birds by the mid-1990s. In 1995, the roost site underwent changes and also the local refuse tip was closed. The roost shifted promptly 1 km to the nearby Doubrava industrial pond, where sometimes gulls had roosted in the past. The members of this roost continued to frequent two other refuse tips in the study area (Ostrava and Havíoov) and probably some other sites in nearby Poland (the border is only *c*10 km from the roost). In 1996 and 1997 the roost moved temporarily to an unknown location, making the counts much less precise in those years. Additionally, the roost site could not be

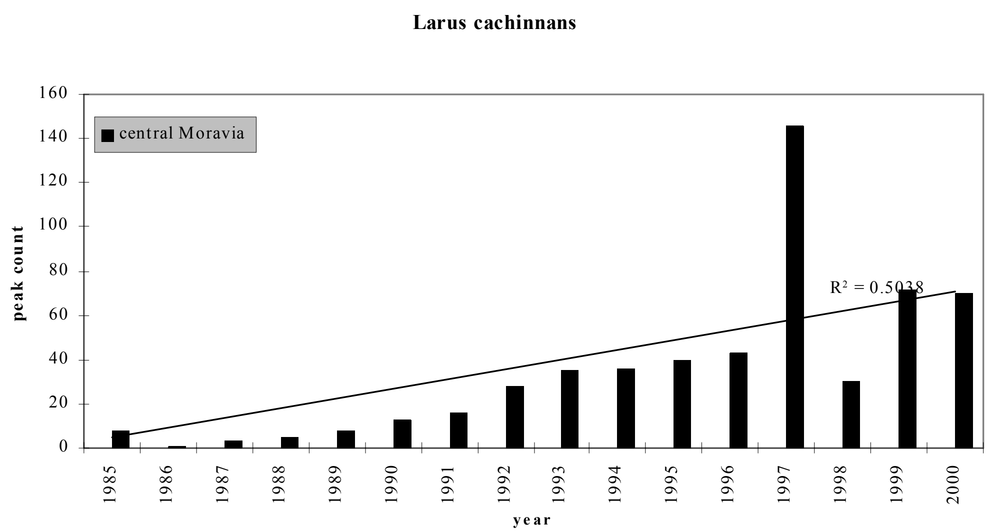


Fig. 3. Population size of *Larus cachinnans* between 1985-2000.

checked in 1999, and so information gull numbers is lacking for that year. However, the Herring Gull has been using the Doubrava roost regularly since 1998 (on 16 Sep 1998 an unprecedented 1670 birds were recorded there). In 2000, 1270 birds were counted on 10 Aug and 1530 on 19 Oct. During the later years of the study, adult birds have outnumbered immatures and juveniles (first- and second-year birds comprise only 20%), a complete contrast to the circumstances at Yellow-legged Gull roosts.

The Herring Gull is now a well-established species in northern Moravia, which it seems to use as a post-breeding moult locale. Larger flocks arrive from late May to June. Numbers build during the summer to peak from late August to September and October. During November, when the flight feather moult has just finished, most birds leave the area and only a small number attempt to overwinter. Such a large assemblage of Herring Gulls occurs only in this area of the Czech Republic; the species is uncommon or even rare elsewhere.

**Yellow-legged Gull *Larus cachinnans***

As for Herring Gull, the results of the Yellow-legged Gull show a large difference between the two study areas, but in the opposite sense. In central Moravia (Fig. 3) the Yellow-legged Gull occurred for the first time in the mid-1980s and quickly established the phenomenon of regular late-autumn 'invasions', which takes place from the first days of November onwards. Birds arrive and stay until it freezes, but in mild winters they attempt to overwinter. Until the mid-1990s, numbers present varied between 20 and 40 birds, but on 9 Nov 1997, an unprecedented flock of 146 birds was recorded. 1998 saw only a small invasion, but over 60 birds were recorded in 1999 and in 2000. More recently, in Feb 2001, up to 124 birds frequented the Tovaeov fishponds. Never before had so many Yellow-legged Gulls been recorded during winter, although some small parties had been observed in Common Gull *L. canus* roosts.

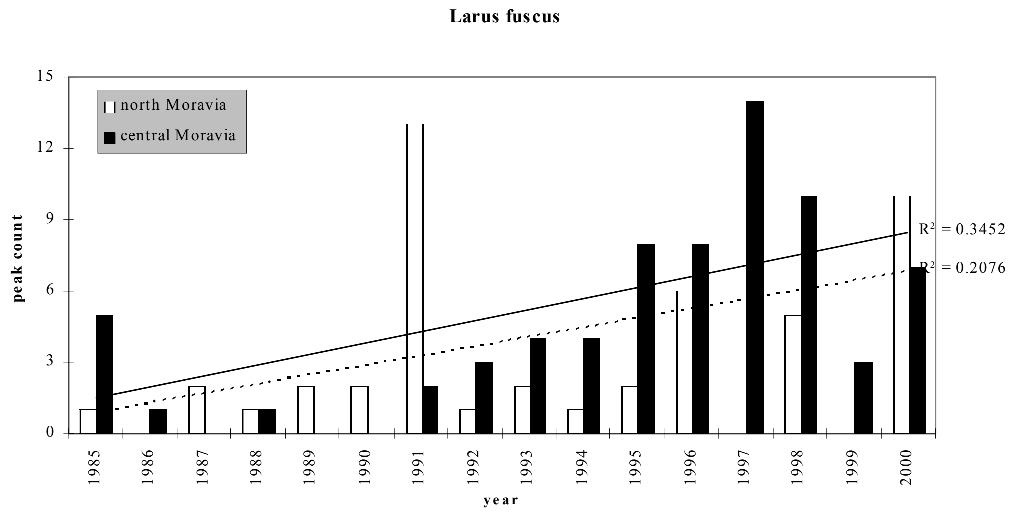


Fig. 4. Population size of *Larus fuscus* between 1985-2000.

During the later years of the study, the Yellow-legged Gull has established itself in central Moravia as a regular autumn and spring migrant, some of which overwinter. It is the earliest gull migrant, the first birds arriving sometimes as early as February, but mainly in March. - This timing seems connected to the species wintering in central and western Europe. Between April and October it is still uncommon, but during late October or early November, many birds arrive, remaining until December (a few stay until January or overwinter). Although there are several refuse tips in that area of central Moravia, the Yellow-legged Gull prefers fishponds and commonly scavenges dead or catches injured fish, often associating with Great Cormorant *Phalacrocorax carbo*.

The situation of the Yellow-legged Gull in northern Moravia is much more complicated. During the study, Yellow-legged Gull occurrence was obscured partly by the increased occurrence of Herring Gulls, but it seemed obvious that only a small number of Yellow-legged Gulls used this area, the largest flocks comprising 10 to 30 birds. However, in November 2000, after the Herring Gull flock apparently had moved to a roost at a small lake near Karvina, it transpired that these gulls were actually new to the area and most of them were Yellow-legged! Most Herring Gulls, of course, have left the area at that time of year. On 24 Nov 2000, 984 Yellow-legged Gulls were counted at the new roost. About 50% of them were first- or second-year birds, distinctly different proportions from those noted for Herring Gull. The occurrence of so many Yellow-legged Gulls in northern Moravia is an abrupt departure from the former pattern, and so much further study is needed to establish whether or not this change is part of a trend.

Preliminary taxonomic examination of the Yellow-legged Gulls in the two study areas reveal that almost all the recorded birds are *L. c. cachinnans*, the Black Sea-Caspian subspecies. Only very few (mainly those recorded in summer) are *L. c.*

*michahellis*, the 'Mediterranean' subspecies. The composition of the two subspecies in the study areas is not typical of elsewhere in the Czech Republic (even southern Moravia), where *michahellis* predominates.

**Lesser Black-backed Gull *Larus fuscus***

Lesser Black-backed Gull, although still not numerous, is now a regular spring and autumn migrant. During the study, a gradual, but obvious increase in numbers of migrating birds has been recorded. Until the late 1980s, this species was very rare and uncommon, only single birds being recorded. However, since the early 1990s small flocks have been recorded more often, becoming regular by the late 1990s. Both study areas have shown the same pattern (Fig 4). Maximum numbers usually are of between 7 and 10 birds. On 14 Apr 1997, 14 birds (including 3 first-winters) were recorded at the Šumvald fishpond.

There is (in contrast to the two above species) a difference between the spring and autumn migration. In spring, very fast movement is typical, birds seldom staying at one site for more than a day; a 5-minute stay is not untypical! In March, but mainly in April, the Lesser Black-backed Gull migrates in single species flocks and does not mix with other large gulls. In autumn, however, mainly from September and November, birds often stay for a long time (up to a month), associating with Herring Gull in northern Moravian roost or with Yellow-legged Gull in central Moravian roosts. The Lesser Black-backed Gull prefers natural habitats and does not visit refuse tips as often as Herring Gull.

## 3. Conclusion

From the collected data (Figs 2-4) it is obvious that the trend of increasing numbers continues for all three studied species, without any significant changes or apparent 'saturation' of numbers occurring. If we take into account recent and unprecedented events such as the new Yellow-legged Gull roost (almost 1000 birds) near Karviná, the overwintering of over 100 Yellow-legged Gulls in central Moravia, the autumn roost of 1500 to 1700 Herring Gulls at Doubrava (small flocks of *c*10 Lesser Black-backed Gulls amongst them), it is clear that the status of these three species in the Czech Republic (or even in inland Europe) will continue to change, and further increases in numbers can be expected.

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# Waterbird-monitoring in the reconstructed habitats of Lake Fertő, northwest Hungary

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## 1. Introduction

Lake Fertő is divided by Hungarian-Austrian national boundary. In the past, the lake and the swamp of the Hanság comprised a single water system. After all the drainage works, reed has extended into the lake basin. At present, 88% of the Hungarian lake water surface is covered by reed, resulting in a reedbelt 2 to 6 km wide separating open water from the lakeshore. As a consequence, the original habitats of the lakeshore-zone have changed. Habitat diversity has decreased, and it has become more difficult to measure and monitor migratory birds.

One of the most important aims of the Fertő-Hanság National Park management now is to restore those bird habitats lost during the last 150 years. Work on this aim began in 1989, and an early result has been the return of same breeding species that had become extinct locally decades ago. Furthermore, the existence of new 200 ha artificially-inundated areas has played a significant role in the reappearance of migratory waterbirds.

### Restoration work so far

There are four territories within the project, each territory being flooded by saline lake-water in a yearly pattern whose initiation and duration depend on the respective conservation management activities they require, activities such as grazing, mowing and reed-cutting that are important to regulate the long-term reed distribution.

There are highly important breeding bird species in the project area, such as Red-crested Pochard *Netta rufina*, Ferruginous Duck *Aythya nyroca*, Avocet *Recurvirostra avosetta*, Black-winged Stilt *Himantopus himantopus*, Kentish Plover *Charadrius alexandrinus* and Mediterrane-an Gull *Larus melanocephalus*. Restoration work has to take the needs of these breeding species into account. Within the project framework, we are examining the how the area functions with respect to the migration process of the most typical waterbirds. To a great extent, the restoration activity provides good opportunity to follow and model the function the lake as a whole plays in the migration process, because the massive extent of the reed coverage of the present lake basin prevents any examination of the present ecosystem by any other methods. In our monitoring project, we trace the population changes of 62 bird species by weekly counts and estimates. It is not possible to monitor rarer species by our methods, nor can we make reliable estimates of the vast flocks of wild geese.

## 2. Interim results

The number of species covered by the project reaches a maximum value by late April and early May (39-42 species; Figs 1a-1c). The total number of birds concerned in the project is at its highest from mid-October until mid-November (8-13 000 birds; Figs 2a-2c). It should be noted that the numbers of wild geese are equivalent in the same autumn period.

The commonest species in the area are Common Teal *Anas crecca*, Northern Shoveler *A. clypeata*, Mallard *A. platyrhynchos*), Northern Lapwing *Vanellus vanellus* and Ruff *Philomachus pugnax*. Endangered (both in Hungary and Europe) and rare species such as Eurasian Spoonbill *Platalea leucorodia*, White-tailed Eagle *Haliaeetus albicilla* and

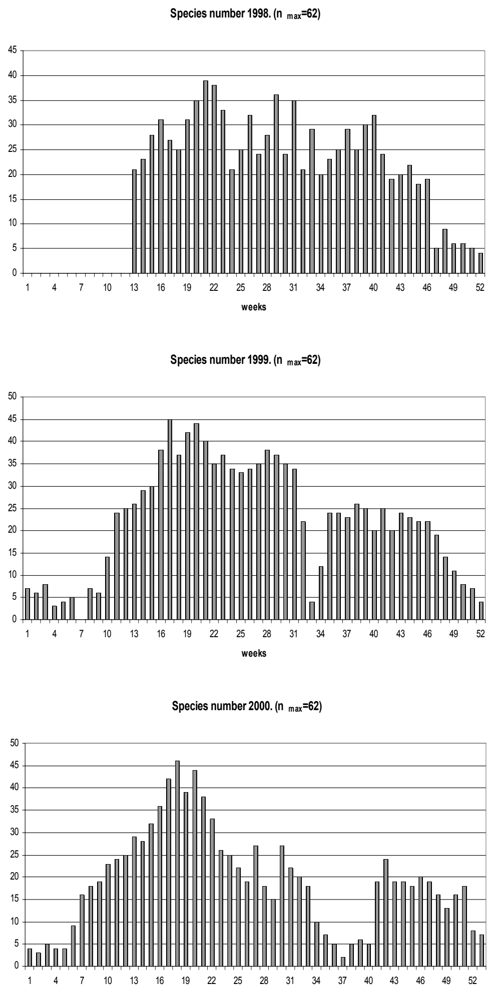


Fig. 1a-1c. The number of species counted weekly in the study area in 1998, 1999 and 2000.

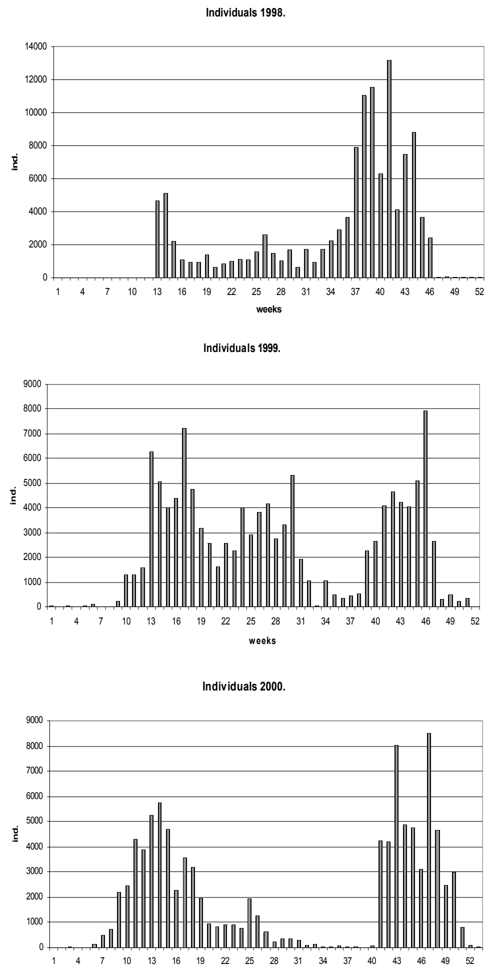


Fig. 2a-2c. The number of individual birds counted weekly in the study area in 1998, 1999 and 2000.

Ferruginous Duck are included in the monitoring project. However, our observations include some critically endangered species, such as Lesser White-fronted Goose *Anser erythropus* and Red-breasted Goose.

The first item of Hanság wetland-reconstruction area (*c*450 ha in size) was completed in 2001, thus increasing the importance of the region to migrant wetland birds. The same monitoring methods will be extended over more of the Hanság in the future, and the project will be enlarged by a joint Austro-Hungarian project to count the wild geese.

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# No Chance of European Monitoring of Raptors and Owls?

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## 1. The Project

The Monitoring of Raptors and Owls in Europe Project was founded in 1988 in Germany at the Martin-Luther-University in Halle and has been carried out continuously since then. Any ornithologist who has investigated a fixed study plot over a period of at least 2 years for its breeding population of raptors and owls can participate. The project also allows studies of reproductive success to be undertaken. Results are entered into form sheets and are sent to the project office in Halle once a year. The magazine 'Annual Report of Raptors and Owls in Europe' is published as a form of feedback for the participants and the scientific community. Detailed descriptions on the Project's development have been published (Gedeon 1994, Stubbe *et al*. 1996, Mammen & Stubbe 2000).

The project aims to obtain comprehensive data on population development of all raptor and owl species in Europe. On one hand, the database provides the basis for protection programmes, and on the other, it demonstrates that changes in the populations of the commoner species can be indicators of environmental changes that remain otherwise undetected.

## 2. The Problem

Although the project comprises data from all over Europe on raptors and owls, after 13 years the main source and focus of the data still lie in Germany, where the project originated. Fig. 1 shows the spread of study plots in Europe. Fig. 2 shows the proportion of registered owl and raptor territories in Germany in comparison those in other countries. Therefore, comprehensive data on owl and raptor population development exists only for Germany and not for other countries, let alone Europe.

## 3. Proposed Solutions

There are a number of reasons why so few co-workers from other states are participating in the project (these reasons also apply in Germany, but to a lesser degree).

1. Not all ornithologists able to participate know about the project. Better publicity and advertising is therefore necessary, possibly in the form of appeals for collaboration placed in national ornithological magazines in different countries. The publication of the Project's results in English more often would certainly help. Multilingual leaflets would be another option.

Fig. 1. Location of study plots for the Monitoring of Raptors and Owls in Europe.

2. Some countries lack sufficiently knowledgeable ornithologists, a difficulty encountered often in European ornithological atlas and monitoring projects. An obvious medium-term solution is for national BirdLife Partners to recruit ornithologists from biology graduates and from literate or informed public, so that knowledge gained on projects could be passed on to others at a local level. Furthermore, ornithological societies should be encouraged to approach and cooperate with the national BirdLife partners.
3. Some ornithologists are reluctant to send their data abroad. This reluctance may stem from a lack of trust or of personal contacts, where they imagine that those who process their data will make errors or will misinterpret the information. Furthermore, the lack of trust may lie in their uncertainty as to how copyright is handled in other countries. We should make it clear to all that our Project's Annual Report, in English and German, is sent free to all Project collaborators. The Report contains the addresses of all contributing collaborators, and readers can use a simple number key to discover whose data has been used. The copyright of course remains with the ornithologist who has collected the data. Although pro-active publicity would help diminish suspicion and mistrust, it needs to be augmented in many countries by organising a network of regional and national fieldwork coordinators who would also

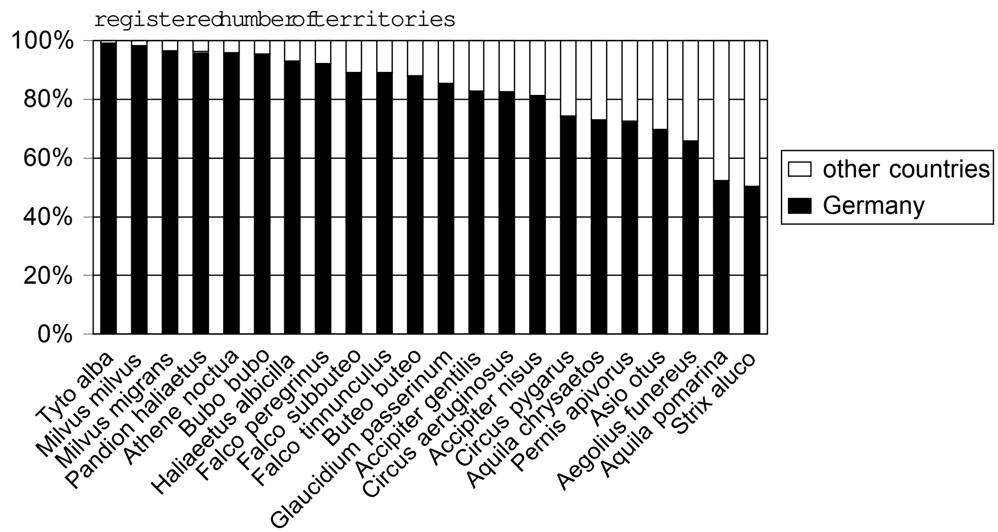


Fig. 2. Proportion of the registered number of territories for selected species in Germany in comparison to those in other states.

collect and collate the data for onwards transmission.

4. For some ornithologists, voluntary work is too expensive of time and money (*e.g.* fieldwork, office work and postage). Although the manhours spent on monitoring study plots cannot be subsidised in general, some form of monetary compensation for eastern European ornithologists would at least not leave them out of pocket for mailing costs, which can be disproportionate to their total income, and thus encourage their participation.
5. Some countries have their own comprehensive raptor and owl monitoring programmes, yet do not participate in the European monitoring programme. The German Project should encourage and invite these countries to participate in the European monitoring programme in a coordinated fashion. Once uniform evaluation methods had been agreed, such as using the same indices, carrying out bilateral or multilateral comparisons would become feasible, a significant step towards a genuine pan-European raptor and owl monitoring programme. Ornithologists who already send their data to national coordination offices that co-operate with the Monitoring of Raptors and Owls in Europe Project would not have to submit their data twice.
6. There is a perception that the Monitoring of Raptors and Owls in Europe isn't a genuine monitoring programme. To challenge this perception, the results achieved by the Project should be disseminated widely and frequently in summary form, the recipients having the opportunity to question all aspects of the Project as feedback. Of course, the Project itself must be subject to reappraisal as an integral part of its development. The formal results from Project should be published in international scientific magazines on a regular basis.

The Monitoring of Raptors and Owls in

Europe Project could be a useful complement to the national common bird census monitoring programmes, because they scarcely consider raptors and owls at all due to these species' relatively low population density. We believe our Monitoring Project offers the opportunity to be representative of studies and conclusions obtained throughout Europe and can therefore become a genuine Euro-Monitoring project. The Projects infrastructure (*e.g.* database and methods of data evaluation) is a sound basis for it.

*Acknowledgements*. We would like to thank all the collaborators from over 17 European countries (so far) for their devoted work. Thanks also to Heike Bose and David Bird who translated the manuscript.

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# Distribution of Little Owl *Athene noctua* and Barn Owl *Tyto alba* in the Zamość Region (SE Poland) in the light of atlas studies

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## 1. Introduction

Although Little Owl *Athene noctua* and Barn Owl *Tyto alba* are known as relatively uncommon, their actual distribution in Poland is poorly known, as emphasised by Ruprecht & Szwagrzak (1988) and Tomiałojć (1990). In particular, little is known about the factors influencing the distribution in Poland of these two owl species; such knowledge is vital for their protection. Furthermore, there is only fragmentary understanding of their population trends in Poland. The purpose of this paper was to analyse the data collected in field studies in the Zamość Region (SE Poland) up to the late 1990s concerning their occurrence.

## 2. Study area and methods

The Zamość region (6980 km$^2$) of SE Poland is typically agricultural in character. About 80% of that area is arable land and 20% is forest. Meadows and pastures comprise some 11% of the region (Anonymous 1996). Two kinds of landscape resulting from agricultural use can be distinguished in the study area. First, the western and central part (except the wooded Roztocze Upland) is a mosaic of small private farm plots separated sometimes by coppices, and the predominant pattern of habitation is of close or linked settlements along the watercourses. Second, the SE Zamość Region is covered largely by monoculture farming where scattered former government farms comprise islands of habitation.

By the late 1980s, the Zamość Region agriculture was in deep recession, manifested by an extremely low level of animal and plant production. In the 1990s the number of cattle and sheep reduced below the levels of the 1970s and 1980s. There was a concomitant reduction in hay production and in the use of grassland as pasture. The result was the appearance of unmanaged land and of large reedy areas in river valleys.

The field studies covered the 1995-1999 period. They consisted of control studies of places suitable for Little and Barn Owls to breed. In all, over 1000 potential nest sites were examined, such as churches, storehouses, schools, dovecots, windmills, barns, sheds, palaces, abandoned buildings, as well as about 200 places such as orchards, alleys of lime (*Tilia* sp) trees and established parks. Much of the study was undertaken in 1995, when some 146 churches were controlled. From 1997-1999, the studies concentrated on former government farm

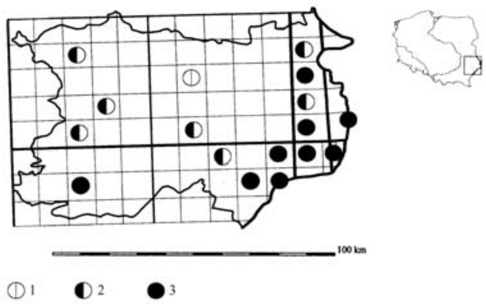


Fig. 1. Distribution of Little Owl *Athene noctua* in the Zamość Region.
1. Pellets, feathers or 1 adult present.
2. Two adults present, voices heard or territorial behaviour of adults observed.
3. Eggs, live or dead nestlings found or juveniles observed.

areas. A large number of sites, particularly in cities, were identified through the use of tape lures, using the methodology of Domaszewicz *et al.* (1984). Following the principles of other Polish atlas models (Ruprecht & Szwagrzak 1988), our map projection was the Universal Transverse Mercator (UTM) in 10×10 km format.

## 3. Results

*Little Owl*. Little Owl strongly avoids wooded areas, and so it was not present in the abundantly wooded areas of Central and Southern Roztocze (Fig. 1). In the western and central Zamość Region, its sites reflected the intensiveness of animal farming of the 1970s and 1980s. Here, the agricultural landscape dominated by small farms, a total of 49 sites of Little Owl was recorded. There, owls of this species populated the pastures and regularly mown waterside meadows of wide valleys of the rivers Bug, Wieprz, Por, Łabuńka and Wolica. Encroachment by headed willow *Salix* sp. is common. However, in this region the Little Owl's preferred nesting sites were in-use or abandoned cowsheds, barns, lofts, drying buildings and houses. Only 2 breeding sites were found in headed willow and apple (*Malus* sp.) trees. Other sites were found in this area of small farms on the outskirts of small villages that lay in a mosaic of fields, meadows, pastures and large orchards that were often grazed. At these latter sites, the Little Owl nested in church towers and attics. Low buildings such as cowsheds, blocks of flats and schools were rarely used for nesting. During these studies, evidence such as live or dead juveniles, an adult pair or fresh pellets were found in only 8 churches (comprising only 5.5% (N=146) of all examined sites in the entire Zamość Region).

In the landscape dominated by monocultures (the large, former government farms comprising *c*20% of the studied region), some 30 sites were found, all associated with the isolated farms. The physical farm building layout and structures are characterised by plentiful accessible nesting places. Inn the past, such places were also suitable for perching and foraging to seek food, because intensive animal farming in this area was beneficial to the Little Owl's prey. At many of these locations (*c*50%), the Little Owl coexists with Barn Owl. In this region the Little Owl breeds in the cities. 25 territories were recorded in Biłgoraj, Hrubieszów, Szczebrzeszyn, Tomaszów Lubelski and Zamość. The species bred in blocks of flats that had plentiful ventilation ducts and were surrounded with regularly mown grass areas, but it was not found in city parks.

*Barn Owl*. This species is more numerous than Little Owl in the Zamość Region. 133 Barn Owl sites were found in the 1995-1999 period (Fig. 2). 109 of these occurred in the mosaic agriculture landscape (small farms) of the western and central Zamość regions. It populated village and settlement ribbons that align with the watercourses of wide river valleys whose watersides are characterised by mown meadows and pastures amid a network of drainage ditches. Other breeding sites were found in localities beyond the river valleys. In the southeast Zamość Region of predominant monoculture farming, the Barn Owl uses the farm building structures for nesting and roosting (N=24 sites), particularly ventilation openings in blocks of flats, attics, cowshed ventilation ducts and barn revetment pillars.

In the agriculture landscape of the western and central Zamość Region, attics and church towers were the main nesting places and diurnal roosts, but others used such as ventilation ducts in village schools and barn revetment pillars. Of churches and orthodox churches accessible to owls (N=58) in the Zamość Region, the Barn Owl occupied 37 (63.8%), brick-built churches being preferred. Rather like Little Owl, Barn Owl is rarely found in the wooded areas of Roztocze. Barn Owl too, steadily populates cities (Biłgoraj, Hrubieszów, Szczebrzeszyn, Tomaszów Lubelski and Zwierzyniec) of the Zamość Region, using attics, church towers and ventilation ducts in blocks of flats or school garrets (Zamość).

## 4. Discussion

The results obtained confirmed the tendencies of the studied species to avoid densely forested upland areas (Exo 1992, Manez 1994, Génot *et al.* 1997, Osieck & Shawyer 1997). Many papers have indicated a close relationship between the occurrence of Little (Exo 1992, Vogrin 1997) or Barn Owl (Colvin 1995) and agricultural trends.

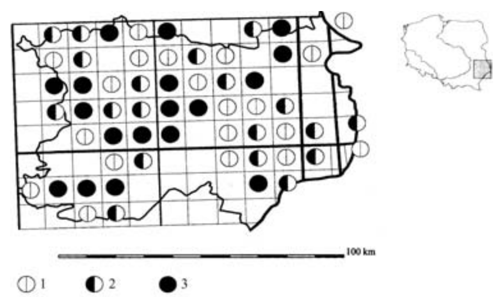


Fig. 2. Distribution of Barn Owl *Tyto alba* in the Zamość Region.
1. Pellets, feathers or 1 adult present.
2. Two adults present, voices heard, or territorial behaviour of adults observed.
3. Eggs, live or dead nestlings found or juveniles observed.

Little Owl occurrence is strongly linked with the abundance of mown or grazed grassland, where the vegetation remains below 15cm in height. Of special importance here are pastures whose boundaries are overgrown by headed willows (Dombrowski *et al.* 1991, Exo 1992). However, in the SE Zamość Region, those habitats populated by Little Owls differ markedly from those in Central Europe considered optimal for this species. It is also worth noting the numerous cases of sympatric occurrence of Little Owl and Barn Owl in the edifices of the former government farms, a circumstance that is favoured firstly by the concentration of accessible sites for nesting in a small area and secondly by the different food preferences of the two species (Mikkola 1983).

Both of the studied owl species require grassy areas lacking dense and high plant cover. This circumstance is guaranteed by regular mowing or grazing, and it allows

Barn Owl to track prey through hearing it move or call (Konishi 1973) and Little Owl to prospect for prey on the ground, because the vegetation is short enough to permit it to run and hop. Therefore, the distinct deepening of the recession of agriculture in Poland since the 1990s may reduce inadvertently the population of both species. The decline in grassland farming and the reduction in the numbers of cattle and sheep will more and more increase the areas of unmanaged land dominated by tall vegetation or reeds.

The accumulation of arable land and the increase in mechanical cultivation may lead to the mosaic agricultural landscape disappearing from the western and central Zamość Region and its replacement by monocultures as has happened in the southeast of the region. In turn, this would cause a decrease in the biodiversity of small mammals, leading to a population inadequate in size to buffer the cyclic fluctuations of the common vole *Microtus arvalis*, posing a distinct risk of starvation for the Barn Owl (Shawyer 1994).

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Vogrin M. 1997. Little Owl (*Athene noctua*): a highly endangered species in NE Slovenia. – Buteo 9: 99-102.

# The occurrence of the Barn Owl *Tyto alba* in sacred buildings in central-eastern Poland

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## 1. Introduction

The occurrence of Barn Owl *Tyto alba* is strongly connected with the presence of buildings with suitable apertures for nesting or roosting (de Bruijn 1994, Shawyer 1994). In Poland this species breeds mainly in tall sacred buildings (Kopij 1990, Kitowski 1999). The main reason for the Barn Owl population decline is loss of suitable breeding sites. In recent years this process has accelerated because attics and towers have been renovated, reducing Barn Owl access (Kitowski 1999). The aims of this study were to determine changes in Barn Owl numbers in sacred buildings and to specify nest site selection preferences.

## 2. Methods and study area

A total of 95 sites were surveyed in 1989-1992 and 120 in 2000 (95 were the same in both these periods). Freestanding belfries and church towers and attics were searched (a total of 113 sacred buildings in 1989-1992 and 152 in 2000). A breeding site was defined as a building where birds or fresh pellets had been observed. The sacred buildings that underwent control surveys were in small towns and villages in farmland of the Mazovia Lowland, an area where arable lands and meadows (68%) were the predominant habitat; woods comprised 22% and built-up areas 8%.

## 3. Results and discussion

The Barn Owl only occasionally occupies belfries, and so changes in numbers were analysed only for church sites (Tab. 1). In the first study period, the Barn Owl occupied 53 churches (59%) out of the 90 surveyed. In 2000, only 36 churches (31%) were occupied out of 115 surveyed. The number of churches with apertures suit-

Tab. 1. Preferences of the Barn Owl in occupation of sacred buildings in central-eastern Poland.

| Years | 1989-1992 | | | | 2000 | | | |
|---|---|---|---|---|---|---|---|---|
| | Sacred Buildings | | Breeding sites | | Sacred Buildings | | Breeding sites | |
| Sacred building types | N | % | N | % | N | % | N | % |
| Brick churches | 77 | 68.1 | 47 | 85.5 | 99 | 65.1 | 32 | 80.0 |
| Wooden churches | 13 | 11.6 | 6 | 10.9 | 16 | 10.5 | 4 | 10.0 |
| Belfries | 23 | 20.3 | 2 | 3.6 | 37 | 24.4 | 4 | 10.0 |
| Totals | 113 | 100.0 | 55 | 100.0 | 152 | 100.0 | 40 | 100.0 |

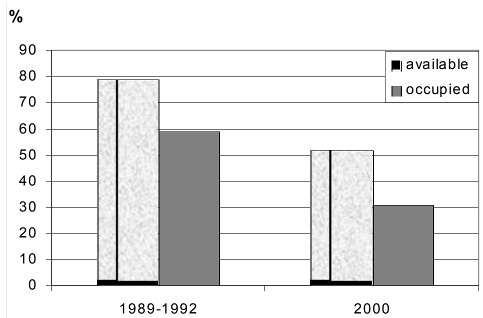


Fig. 1. Proportion of available and occupied Barn Owl breeding sites in central-eastern Poland.

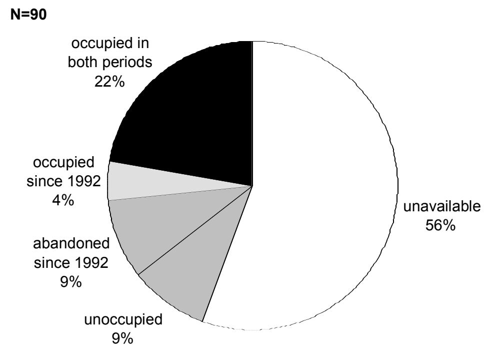


Fig. 2. Barn Owl's choice of breeding sites in sacred buildings in central-eastern Poland.

able for Barn Owl decreased significantly during the study period: 79% were available in 1989-1992, but only 52% were available in 2000 (Fig. 1). In 10 years, this species had disappeared from 31 churches (of the 53 occupied in 1989-1992), a decline of 58.5%. In 23 of the 31 cases (74.2%), the reason was that the aperture openings had been blocked up. When apertures were made in five churches, Barn Owls appeared in all of them. Of the 90 churches surveyed in both periods, 40 remained available to the Barn Owl throughout. Over the 10-year period, the Barn Owl remained in 20 churches, abandoned 8, colonised 4, and failed to occupy 8 (Fig. 2).

In both periods the Barn Owl preferred brick churches, but clearly avoided free-standing belfries (Tab. 1). The low occupation rate of wooden churches and free-standing belfries probably was due to the low height of these buildings and to the concomitant reduced security. A decrease in Barn Owl numbers has been observed in most of Europe (Heath *et al*. 2000). Across Europe, the main threats to the Barn Owl are the decreasing areas of grasslands (preferred hunting habitat) as result of agriculture intensification, urbanisation and transport development (Colvin *et al*. 1984, Colvin 1986, Newton *et al*. 1991, Bunn *et al*. 1992, de Bruijn 1994, Shawyer 1994). In central-eastern Poland, the main factor responsible for the disappearance of the Barn Owl from churches is the sealing of building aperture openings in order to eliminate breeding Jackdaw *Corvus monedula*. To afford safe breeding sites for the Barn Owl, it is necessary to re-open the apertures in churches and to erect nest boxes in suitable habitat that contains adequate year-round food for the species.

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# The monitoring of Little Owl *Athene noctua* in Chełm (SE Poland) in 1998-2000

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## 1. Introduction

For many years in parts of Europe, Little Owl *Athene noctua* populations have declined, sometimes to extinction (Manez 1994, Genot *et al*. 1997). Little Owl has also disappeared from territories in Poland, particularly in the west where a sharp population decline has been recorded in agricultural landscapes and cities (Tomiałojć 1990, Dyrcz *et al*. 1991, Jermaczek *et al*. 1995, Bednorz *et al*. 2000).

The first data concerning Little Owl numbers in 20 cities in Lublin district (SE Poland) were obtained in the 1990s. Chełm was found to have the highest population and density of this species (Grzywaczewski & Kitowski 2000a). This population was the ideal choice for monitoring studies, the purpose of which was to estimate the numbers, distribution and nesting preferences of the species.

## 2. Study area and methods

Chełm (51°08'N, 23°30'E) is a medium-sized city in southeast Poland (Fig. 1), 35.7 km$^2$ in area, and has some 70 000 inhabitants. The chalk hills on which it is built ranges from 178-232 m asl in height (Kondracki 2000). Its climate is continental, and the mean maximum temperature (24.2°C) is amongst the highest in Poland (Kaszewski *et al*. 1995).

The city landscape is characterised by a high percentage (38%) of arable land and habitation (36%), an inefficiently developed road system (11%), small woods and bushy areas (7%), and industrial estates (8%). The arable land includes cultivated areas, some larger than 10 ha, meadows, smallish orchards and vegetable gardens. The accommodation comprises single-storey houses or blocks of flats, which are mostly limited to four storeys due to the calcareous chalk bedrock. The flats are surrounded by mown or trodden

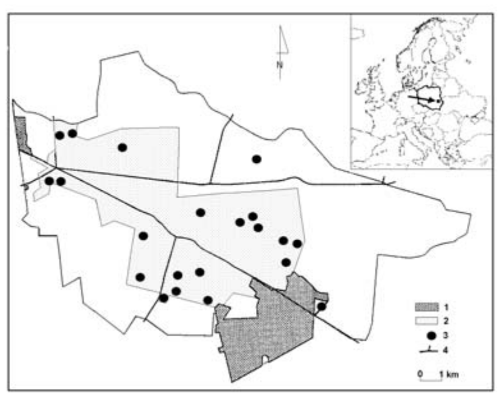

Fig. 1. Distribution of Little Owl *Athene noctua* in Chełm (SE Poland).
1. Forests.
2. Single-storey houses and blocks of flats on estates.
3. Little Owl *Athene noctua* territories.
4. Main roads.

grassy squares. Most homes are adjacent to arable land (Chełm City Hall records).

The observations in 1998-2000 were conducted according to a standard recommended method (Domaszewicz *et al*. 1984), the study area being the city of Chełm. We plotted on a 1:10 000 map the detected Little Owl calling and contact points. the resultant network showed the points to be *c*300-400 m from one other. We made two effective nocturnal control surveys of the city during each reproduction season, as were 1-2 additional controls in places where territorial occupancy was uncertain. The study period covered the peak of Little Owl calling activity from the last third of February to the end of April (Exo 1988). Most controls were carried out during the first peak of nocturnal calling activity, which occurs from sunset till about midnight (Exo 1989). We also conducted several additional night-long observations. On windless and rainless nights in suitable periods of high-pressure weather, we used voice stimulation. In addition, we conducted day controls, the purpose of which was to localise territories, particularly those where birds had not reacted to nocturnal voice stimulation. This enabled us to search for other traces of Little Owl presence, such as feathers, pellets, and roosts of young and adult birds.

## 3. Results

In the city area, there were 14-19 Little Owl territories during the studies, giving a territory density of 4.0-5.3/10 km$^2$ (Tab. 1). Of N=21 territories, most (67%) were found to be close to the blocks of flats, while 28% were adjacent to the single-storey houses. Only one territory (5%) was found in the industrial part of the city. However, most territories (81%) were recorded in transitional zones between the areas of human habitation and fields, meadows and gardens. Only four (19%) territories were found in the city centre (Fig. 1), but they were close to gardens, parks and lawns. In the above territories, the Little Owl used holes in ceilings, roofs, chimneys, openings and attics. Despite the presence of numerous trees with holes, we found no nests in the city area parks and woods.

## 4. Discussion

In the 1980s and 90s, Little Owls were counted in a number of Polish cities. Reliable censuses were made in Gliwice (W Poland) (19-20 bp/136 km$^2$, Tomiałojć 1990), Kraśnik (SE Poland) (8 bp/32 km$^2$, Frączek & Szewczyk 2000). For Hrubieszów (7 bp/32.8 km$^2$), Biłgoraj (4 bp/20.8 km$^2$) and 17 other cities and small towns in SE Poland, the breeding density ranged from (0.07-4.5 bp/10 km$^2$ (Grzywaczewski & Kitowski 2000b). In comparison to those figures, the number and density of Little Owls in Chełm are amongst the highest in Poland. The Chełm figures were several times higher than those pertaining to agricultural landscape areas rich in meadows and willows (*Salix spp*) or to extensive orchards considered

Tab. 1. Number and density of Little Owl *Athene noctua* territories in 1998-2000.

| Years | Number of territories | Number of territories/10 km$^2$ |
|---|---|---|
| 1998 | 17-19 territories | 4.8-5.3 |
| 1999 | 14-16 territories | 4.0-4.5 |
| 2000 | 14-16 territories | 4.0-4.5 |

optimal for Little Owl in Poland and Central Europe, where the relevant densities ranged from 0.1-1.5 bp/10 $km^2$ (Jermaczek *et al.* 1990, Dombrowski *et al*. 1991, Fronczak & Dombrowski 1991, Kowalski *et al*. 1991, Vogrin 1997).

Away from cities, in central and western Europe, hollow trees seem to be the preferred breeding sites of Little Owl (Manez 1994, Génot *et al*. 1997). However, in Chełm and other southern Polish cities (Grzywaczewski & Kitowski 2000a, 2000b) the species avoided nesting in hollow trees. In the agricultural environs of Chełm, nesting occurred only in buildings, as it did in Chełm itself, where blocks of flats were particularly favoured (Grzywaczewski 2000). Furthermore, nesting in hollow trees elsewhere in the agricultural landscape of southeast Poland is exceptional (Kitowski & Kisiel 2003). These findings contradict those in Little Owl data gathered in Poland up to the 1980s (summarised by Ruprecht & Szwagrzak,1988), which cite many cases of nesting in hollow trees in parks and municipal cemeteries.

The high numbers of Little Owl in Chełm seem to be influenced by the ease of access to plentiful breeding places in estates of blocks of flats and single-storey buildings. Characteristic of these estates are open spaces comprised of regularly mown or trodden grass, suitable foraging grounds for the owls. The typical calcareous hilly landscape of the city and its surroundings and the relatively warm climate favour dry vegetation cover suitable for Little Owls to hunt, for here such a short-legged species faces none of the difficulties in catching prey that it encounters on short-cut hay grasslands.

Little Owl monitoring will continue, enabling us to investigate the population trends. It will also help us to undertake measures to help protect the species from population decline.

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# Trends in numbers of the Coot (*Fulica atra*) in the Czech Republic in 1988-2000

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## 1. Introduction

Common Coot *Fulica atra* is one of the commonest waterbird species in the Czech Republic. The total size of the breeding population was estimated between 1985 and 1989 as 30 000 to 60 000 bp (breeding pairs) (Šťastný *et al.* 1996). In the 20th century, up to the late 1970s, breeding population numbers increased during most decades (*e.g.* Fiala 1978, Řepa 1987, 1989). During the 1980s and 1990s, a remarkable decline in Coot numbers was recorded (Musil 1995, 1999, Musil *et al.* 2001). However, our present knowledge about the population dynamics of this species is very poor, circumstances that provided the main stimulus for the initiation of an intensive study of Coot ecology and breeding biology, including the annual monitoring of breeding population size. This short paper is aimed at an analysis of the regional pattern in trends in Coot breeding population numbers.

## 2. Study area

### Fishpond characteristics

In the Czech Republic, artificial lakes and fishponds represent the most suitable breeding habitat for Coots and many other water birds (*e.g.* grebes, geese, and ducks). Fishpond development began in Czech lands mainly during the 14th century. During the 20th century, fishpond ecosystems have changed markedly in term of fish production, biodiversity and water quality, the most remarkable changes in fishpond management being recorded from the 1950s onwards. The documented evidence shows heavy increases of nutrient flow both from the surrounding landscape and from direct fertilization of the waterbodies. The consequence has been that fishponds have changed from being oligotrophic and mesotrophic waterbodies to become eutrophic and hypertrophic in the 1980s and 1990s. In these two decades, the mass of fish stocks has also increased greatly for many decades (from 50 kg/ha in the 1900s to over 500 kg/ha).

The enormous scale of the grazing effect of fish recently seems to be the most important factor affecting benthic and plankton communities, the extent of littoral vegetation and consequently also many other limnological parameters, such as water transparency and chemistry (Pokorný *et al*. 1994, Pokorný & Pechar 2000). The above massive eutrophication also led to the intensive development of littoral vegetation in fishponds. Consequently, water surface area has declined in many fishponds since the 1950s. To counter this


Fig. 1. Location of investigated fishponds regions in the Czech Republic: 1. Plzeň, Tachov. 2. České Budějovice. 3. Třeboň. 4. Jindřichův Hradec. 5. Central Bohemia. 6. East Bohemia. 7. North Moravia. 8. Břeclav.

effect, the commercial fish breeders have carried out large-scale removal of sediment and littoral vegetation.

## 3. Methods

The programme 'Monitoring Waterbird Breeding Populations in the Czech Republic' started in 1988 (Musil 1995, 2000). Annually, volunteers collect data from 460-710 water bodies in some 20-40 varied regions of the country. Data from the 8 most important fishpond regions (České Budějovice, Jindřichův Hradec, Třeboň, Tachov and Plzeň of Central Bohemia, and Břeclav, Central and North Moravia, East Bohemia) were analysed for this paper (see Fig. 1).

Since 1993, the results have been published in annual reports in a bulletin, CSO (Czech Society for Ornithology) News (*e.g.* Musil 1997, 2000). All ponds under study are checked twice during each breeding season, in the second half of May and in the second half of June. In case of Coot, as a territorial species, the higher of the two values was used to express the species abundance (see Musil 1996 for details). Relative changes in breeding population size were expressed by population index calculated as a percentage of the base-year (1988) population size. The statistical importance of the population trends was assessed using the linear correlation coefficient. Principal Component Analysis (PCA) was used for comparison of population trends in fishpond regions (Fig. 3).

## 4. Results

Coot was the most frequently recorded water bird species, occurring on more than

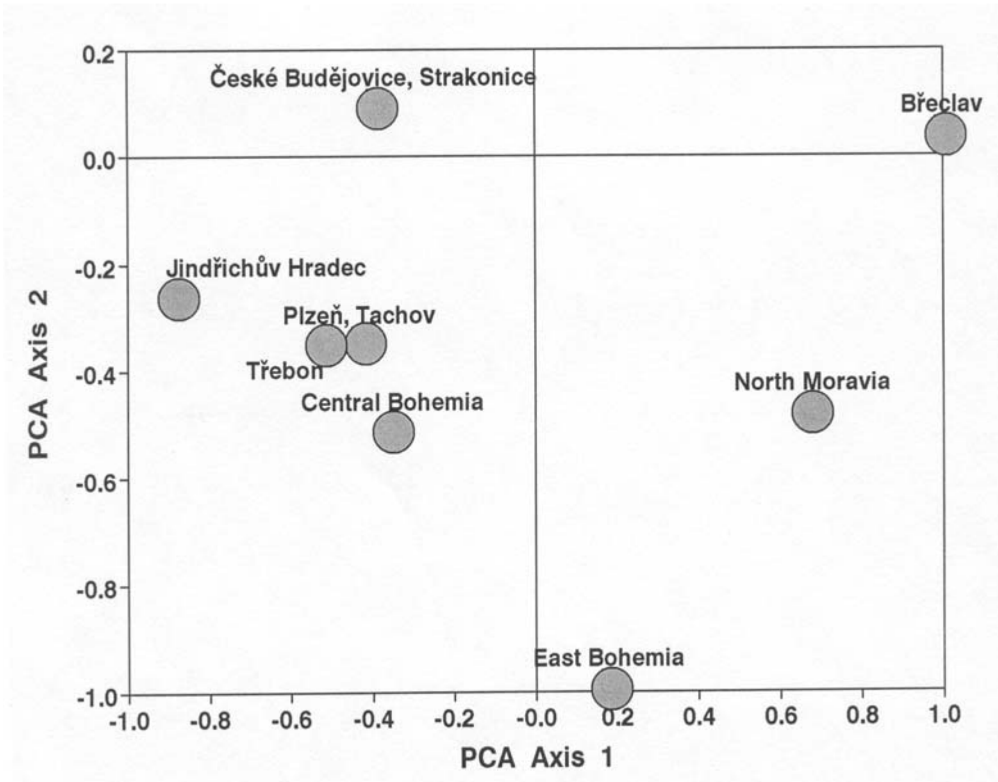


Fig. 3. Principal Component Analysis of inter-region similarity in trends in numbers of breeding population of Coot.

63% of studied water bodies. Since 1989, a significant decrease in Coot numbers has been recorded in most of the fishpond areas (see Fig. 2). The most significant decrease was recorded in South Bohemia, including the regions of České Budějovice (r=-0.870, P<0.001, n=13), Jindřichův Hradec (r=-0.944, P<0.0001, n=13) and Třeboň (r=-0.873, P<0.001, n=13). Moreover, rapid decreases in Coot breeding population numbers were found in Southwest (Tachov and Plzeň) (r=-0.767, P<0.01, n=13) and Central Bohemia (r=-0.709, P<0.01, n=13). Significant increases since were recorded in Břeclav region and in Central and North Moravia. Fluctuations in Coot numbers were recorded in East Bohemia (r=0.078, n.s., n=13).

The similarity of the population trends in particular fishpond regions (PCA, see Fig. 3) reflects the geographical location of these regions. There is a very visible difference between trends in the west (most Bohemian regions) and in the east of the country (Moravia).

## 5. Discussion

The Coot breeding population increasing in many decades of the 20th century but declined very rapidly in the 1980s (Šťastný *et al.* 1996, Musil 1999). These population changes do not correspond with the general population trends recorded in the Western Palearctic (*e.g.* see

Gorban & Stanevičius 1997, Delany *et al.* 1999). Many possible causes of this aberrant decline have been discussed. The following factors are often strongly connected with the intensification of fishpond management, which has affected the entire fishpond plant and animal communities since the 1980s.

1. *Cutting, destruction or degradation of the littoral vegetation*. The littoral veg-

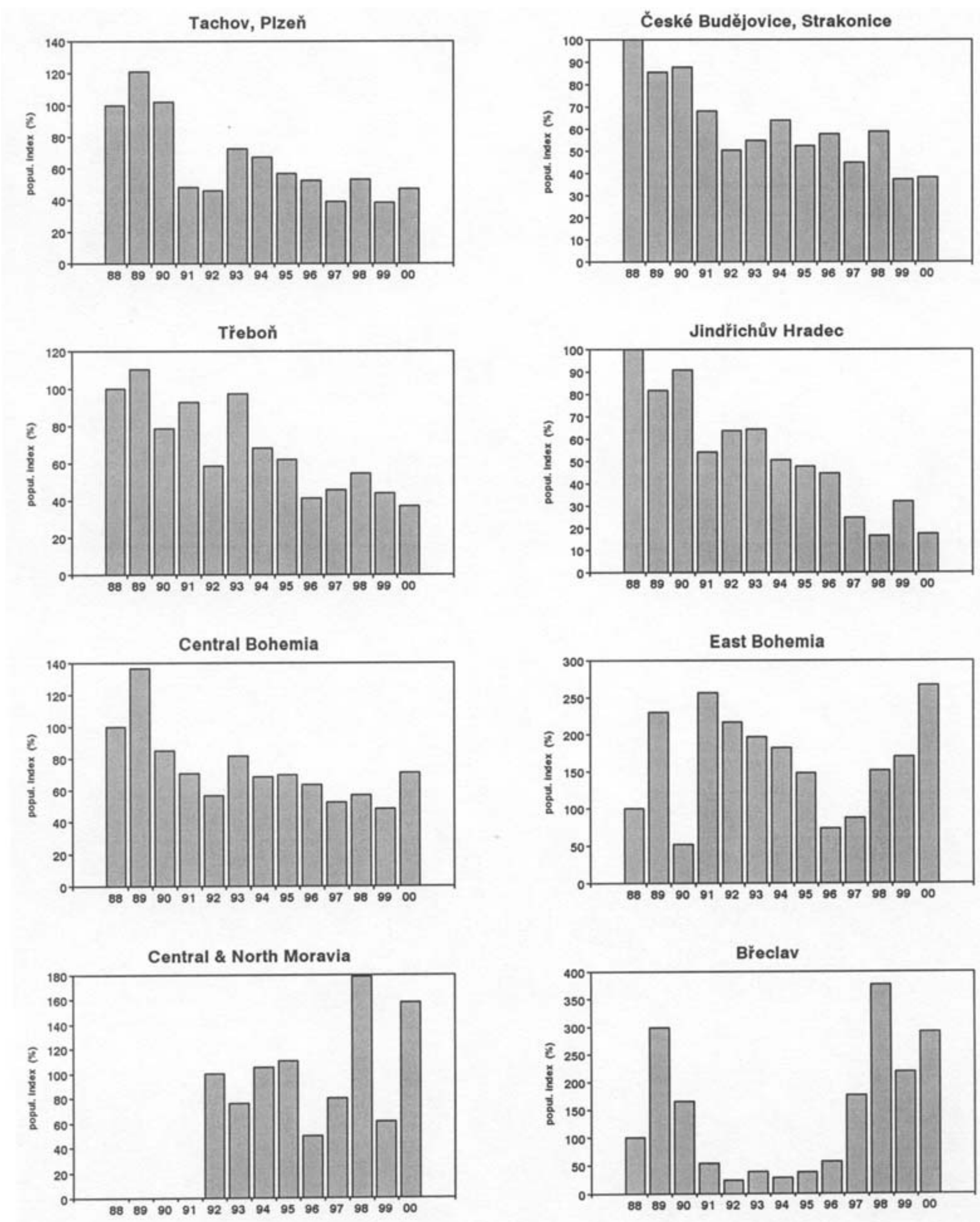


Fig. 2. Breeding population changes of Coot in various regions of the Czech Republic in 1988-2000.

etation mostly comprises *Typha*, *Phragmites* and *Glyceria*, and damage to it causes loss of nest habitat. Because the Coot is territorial, when breeding habitat is in short supply, the weaker pairs often nest in unsuitable habitat or fail to breed. The loss of nest habitat exacerbates the predator pressure on nests and chicks, because so many nests are badly hidden (Corvidae, Marsh Harrier *Circus aeruginosus* and Grey Heron *Ardea cinerea* benefit most).

2. *Overstocking of fish in fishponds*. The dominant fish species is the omnivorous carp *Cyprinus carpio*, and excessively large fish stocks cause food competition between the fish and birds in the brood feeding period (newly hatched chicks are fed invertebrates). Moreover, the presence of large carp has had a negative effect on littoral vegetation quality.
3. *Direct fertilization of water bodies and artificial feeding of fish stocks*. The high nutrient input from the presence of fertilizer (including run-off from agriculture) and the degradation of the fish food (the consequence continuous supply of fish excreta) cause phytoplankton (algae, cyanophytes) to develop, thus decreasing water transparency. The conditions that have caused the reduced food supply for birds are more likely to allow the development of anaerobic bacteria such as *Clostridium botulinum* that infect birds with botulism, thus increasing bird mortality rates.

Although Coot numbers show a decreasing trend in most Czech fishpond regions, increases in numbers were recorded in several regions after 1995, where newly established roosting sites hosted large post-breeding and non-breeding Coot aggregations, especially in Moravia. Improved management of several larger fishponds (protected as part of the National Nature Reserves) could affect this trend, particularly if the actions include diminishing the fish stocks and lowering the water level. The consequences of introducing such management practices would have a positive effect on the development of submerged water vegetation.

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# Changes in Breeding Population Numbers of Kentish Plover *Charadrius alexandrinus* at Atanasovsko Lake

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## Abstract

Atanasovsko Lake is the most important Kentish Plover breeding site in Bulgaria. 141 pairs nested there in 1978, 252 in 1979 and 238 in 1981. During the following years, numbers decreased sharply due to habitat degradation, reaching 39 pairs in 1993 and 65 in 2000. Detailed studies in 2000 showed that the first spring migrants arrived 16 March, nest-building began mid-April and egg-laying after 20 April. The latest nests with eggs were found on 15 July. The decline in numbers of the Kentish Plover breeding population at Atanasovsko Lake are due mainly to narrow watercourses, mostly dikes, becoming overgrown by tall vegetation. To a lesser degree the birds suffer from numerous ground predators, such as red foxes, jackals and feral dogs and cats. Nowadays the Kentish Plover prefers to nest where the lake is being worked, on the board-lined banks that mark out the salt production basins.

## 1. Introduction

The Kentish Plover *Charadrius alexandrinus* is a cosmopolitan species occurring from the southern coast of the North Sea, along the south European shores of the Atlantic, in the Mediterranean, Black and Caspian Seas and round the Arabian Gulf. It also occurs inland in Central Asia, in the east reaching to the Trans-Baikal and Mongolia. In Bulgaria it breeds along the Black Sea coast and round the adjacent lakes, the core breeding population being concentrated mainly at Atanasovsko Lake in East Bulgaria. Atanasovsko is a shallow, hyper-saline lake divided by a motorway into northern and southern parts. The lake, being used for salt-production, is divided into numerous small ponds by a network of dikes and banks. In recent years habitat degradation has led to a serious decline in Kentish Plover numbers.

## 2. Methods and Materials

The material for this work was collected during the 2000 breeding season, from March to July. The lake was searched regularly for nests, which were described and mapped. All existing nearby breeding locations were also included in the analyses.

## 3. Results and Discussion

The earliest information about Kentish Plover breeding at Atanasovsko Lake came from the mid-19th century, when

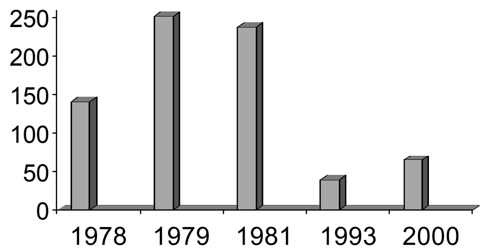


Fig. 1. Number of breeding pairs of Kentish Plover at Atanasovsko Lake by count years.

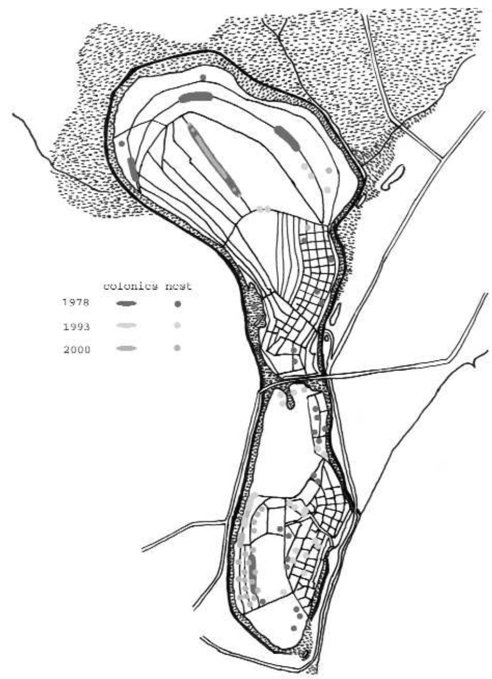


Fig. 2. Colonies and nests of Kentish Plover at Atanasovsko Lake in the years.

Elwes & Buckley (1870) found a nest with 2 eggs in early April 1869. Otmar Reiser (1894) also found the species nesting at Atanasovsko Lake, but at that time most of the breeding population bred on the sea-coast. Subsequently, many other ornithologists recorded the species along the Black Sea coast. A. Darakchiev & D. Nankinov (1979) compiled the first detail counts and maps at Atanasovsko Lake, finding 141 nests between 10 and 20 May 1978. In 1979 there were 252 breeding pairs (bp) and in 1981 238 (Nankinov 1989), these figures being maxima recorded (Fig. 1). A subsequent sharp decline followed, reducing to only 39 bp in 1993, a sixfold reduction (Nankinov 1994).

In 2000 the first return migrants, two individuals, arrived on 16 March. Pair-formation and displaying birds were observed by late March. Nest-building began in mid-April and egg-laying after 20 April. By 27 April there were 9 nests with eggs in the southern part of the lake (6 nests had 1 egg, 1 nest had 2 eggs and 2 nests had 3). After this early start, the breeding season peak was later than previously recorded (10-20 May in 1979 [Darakchiev Nankinov 1979]) most nests being recorded in 2000 between 20 May and 19 June. A total of 38 nests were recorded then (58% of all nests found). In late June the number of nests found gradually decreased; 16 (24%) of the total found. Two late broods were found on 15 July on the northern part of the lake.

In 2000, a total of 65 nests were found at Atanasovsko Lake: 49 nests were in the southern part and 16 in the northern. Some 25 bp comprised 3 small colonies of 5, 10 and 10 bp respectively; 40 bp nested separately. Two colonies (of 5 and 10 bp) were in the southern part of the lake and the other was in the northern part (See Fig. 1 for comparisons between count years.

Fig. 2 compares the positions of nests and colonies of Kentish Plover at Atanasovsko Lake in the years 1978, 1993 (Nankinov, 1994) and in 2000 shows that the previous numerous colonies have almost disappeared, only small colonies and separate pairs remaining. The species' favoured nesting locations of open dikes have been overgrown by vegetation over 1m tall. These dikes now can scarcely be negotiated, and are frequented by red foxes *Vulpes vulpes*, jackals *Canis aureus* and feral dogs and cats. Kentish Plovers no longer breed there. The remaining colonies are on board-lined banks located in the working part of the lake, where the birds are often disturbed. Those dikes and dried-out parts of the ponds that became covered in glasswort *Salicornia europaea* became suitable secondary breeding habitat for Kentish Plover to nest, and are now the favoured breeding locations at Atanasovsko Lake. 37 nests (56.9%) were found in glasswort areas, which afford them concealment from ground and aerial predators. The effectiveness of this camouflage is shown that out of 11 nests (17%/) lost, only 2 were found by roaming feral dogs whereas 9 were flooded. No unfertile or unhatched eggs were found.

## 4. Conclusion

I deduce that the declines in Kentish Plover breeding population numbers at Atanasovsko Lake are due mostly to habitat degradation, namely overgrowing of dikes by tall vegetation. A secondary cause is the presence of numerous ground predators. Nowadays, the Kentish Plover prefers to nest in the working part of the lake. For information on other potential factors influencing these declines in the northern part of the lake, such as a sudden cold spells during incubation and brooding, and disturbances from increasing vehicle movement, other habitat changes and the increase in the presence of humans and cattle, see Dybbro (1970) and Jonsson (1983).

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# Breeding Woodcock *Scolopax rusticola* monitoring in France

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## Abstract

The French breeding Woodcock population has been monitored since 1988, and since 1991, the monitored geographical area has been representative of that population. Monitoring is based on observing roding males, the methodology being random sampling with replacement. The annual results are expressed as a frequency of occurrence. The data collected since 1988 allow us to define very precisely the breeding distribution area of the French Woodcock population. The collected data have shown that the French breeding Woodcock population has been stable from 1991 to 2000.

## 1. Introduction

The Woodcock *Scolopax rusticola* is present all year round in France. The wintering population in France, mostly in the south and west, is augmented significantly by a large proportion of breeders from Scandinavia, north-eastern and central Europe. The established French breeding population is at the southwestern limit of the main European breeding area. This population has been monitored since 1988, the monitored geographical area having been representative of the entire population since 1991.

## 2. Census method

Monitoring is based on evening observations of roding males. Previous research has shown that the roding sites in May and June identify potentially favourable breeding areas (Ferrand unpub; Hirons 1987), the amount of calling increasing with the number of males present (Ferrand 1987). Ferrand (1993) described the basic methodology. France being divided into *départements*, we used a stratified sampling design in each. For each *département* we obtain a number of 1:50 000 maps, in which we define forested sampling units (2×2 centigrades). Each year, samples are obtained randomly for about 10% of the sampling unit population. A defined listening point is estab-

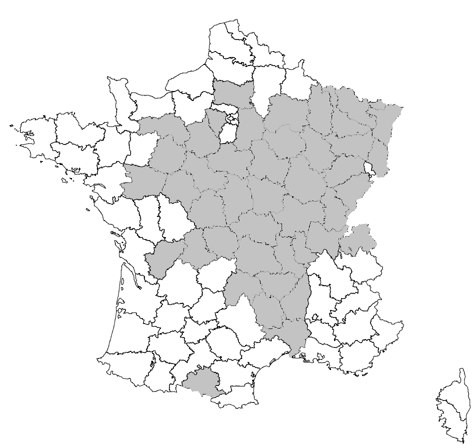

Fig. 1. Monitoring areas of roding woodcocks from 1991 to 2000 (yearly).

lished as close to the centre of the sampling unit as possible. A single visit is made in May or June, when all Woodcock seen and heard are recorded for about 90 minutes (the duration of the evening roding period).

We use a method of random sampling with replacement; *ie* the sample is renewed each year and all members of the sampling population are given an equal chance of being drawn.

## 3. Results

The annual results are first expressed as a frequency of occurrence. Two abundance classes are also defined: 1-4 observations (low abundance) and more than 4 observations (high abundance).

**Population trend**

We can establish population trends from data collected continuously in 42 *départements* since 1991 (Fig. 1). The frequency of occurrence (proportion of sampling units containing at least one Woodcock, seen or heard) is stable ($\chi_1^2$=1.495; P=0.221; Fig. 2). We can also define stability for high and low abundance sampling units ($\chi_1^2$=2.17; P=0.14 and $\chi_1^2$=0.137; P=0.711 respectively; Fig. 2). Our collected data show that the trend in the French breeding Woodcock population has been stable from 1991 to 2000.

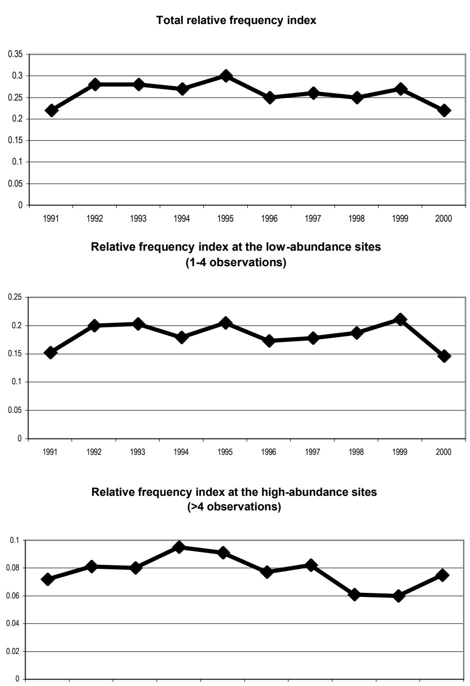


Fig. 2. Trend of the total relative frequency index (proportion of sampling units with at least one seen or heard roding woodcock) and of the low (1-4 observations) and high (>4 observations) abundance sampling units from 1991 to 2000.

Fig. 3. Distribution area of roding males for the period 1988-2000.

*Legend*

Maps with at least one visited sampling unit (listening station)

- Presence
- Absence
- Maps not covered
- Non-participant *départements*

**Distribution**

The data collected since 1988 allow us to define a very precise distribution area for

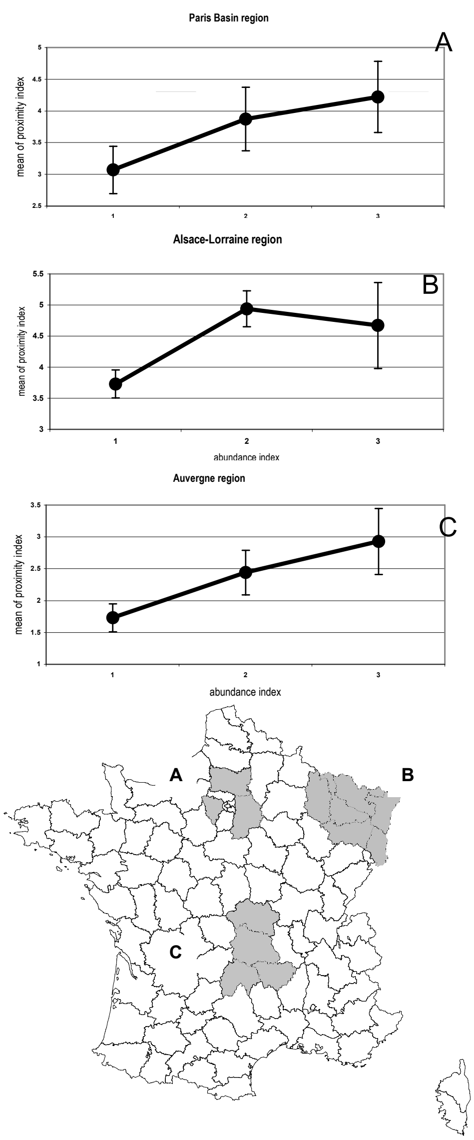


Fig. 4. Relation between the mean of proximity index [value from 0 (isolated sampling unit) to 8 (unit entirely surrounded)] and the abundance index (1 = 0 observation; 2 = 1-4 observations; 3 = >4 observations) for three French regions.

the French breeding Woodcock population. This can be done at two levels, national and regional. The 1:50 000 maps represent the basic unit at national level (Fig. 3) and the forested sampling unit the basic unit at regional level. The northeastern part of the species' distribution, the mountainous regions and the large forests of the Paris Basin comprise the majority of the Woodcock breeding distribution in France.

We can obtain ecological information from this census. For example, an isolation/proximity frequency index can be attributed to a sampling unit according to the number of joined sampling units. The index value varies from 0 (isolated unit) to 8 (unit entirely surrounded). A first analysis of the habitat fragmentation effect (isolation *v* abundance) for three French regions shows that breeding Woodcock is significantly more numerous in large forests (Fig. 4). However, further research is needed to establish this effect more precisely to accord with the forest topography and structure.

## 4. Conclusion

The data collected by this census method - a population trend established through distribution and ecological information - are necessary to ensure management of this game species.

Woodcock censuses have also been carried on in Switzerland since the late 1980s, the census method being identical to that in France, (Estoppey 2001) and in Russia since 1999, their census method being sampling units randomly chosen within 12×12 km quadrats). Now our objective is by coordinated work in other countries to apply our or similar methodology to as much of the Woodcock's European breeding area as possible.

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# Abundance of Four Lark Species in Relation to Portuguese Farming Systems

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## 1. Introduction

Throughout much of Europe, farmland birds have declined more than those of other habitats, because of the abandonment of traditional farming systems and the simplification of remaining agricultural systems, including increased use of external inputs (Tucker & Heath 1994, Baldock 1991, Bignal & McCracken 1996). Traditional low-input agricultural systems survive in some parts of southern Europe, including Portugal, where the main arable region is Alentejo in the south of the country. However, such systems are not currently economically viable.

Tab. 1. Agricultural statistics for the three land-use categories considered in Alentejo, Portugal (source: Cordovil 1993).

| | **Intensive** | **Extensive** | ***Montado*** |
|---|---|---|---|
| Mean farm size (ha) (all farms) | 48 | 161 | 66 |
| Crop area (%) | | | |
| Total annual crops | 81 | 42 | 28 |
| Winter cereals | 45 | 40 | 21 |
| Sunflower | 20 | 0.3 | 0 |
| Forage crops | 6 | 2 | 7 |
| Fallow | 15 | 52 | 66 |
| Perennial crops | 11 | 0.8 | 2 |
| Olive | 10 | 0.8 | 2 |
| Vines | 1 | 0 | 0 |
| Land area per tractor (ha) | 58 | 125 | 194 |
| Livestock (%) | | | |
| Sheep | 68 | 78 | 83 |
| Cattle | 8 | 11 | 7 |
| Pigs | 22 | 8 | 5 |
| Goats | 2 | 3 | 5 |

Larks (Alaudidae) represent a passerine family that is strongly associated with farmland landscapes. In Portugal, the species present include Calandra Lark *Melanocorypha calandra*, Woodlark *Lullula arborea*, Short-toed Lark *Calandrella brachydactyla* and Skylark *Alauda arvensis*. Of these, Woodlark and Short-toed Lark are widely distributed as breeding species in Portugal, while Skylark occurs mainly in the north of the country and Calandra Lark mainly in the southeast (Rufino 1989). Iberian populations of all four species declined in numbers during the 1970s and 1980s and are currently the subject of European conservation concern (Tucker & Heath 1994). This study assesses the abundance of these species in December and April in relation to three arable systems in an agricultural landscape of Baixo Alentejo, southern Portugal.

## 2. Study area

The study area included parts or all of five administrative regions in Baixo Alentejo (Ferreira do Alentejo, Aljustrel, Castro Verde, Ourique and Almodôvar), an area

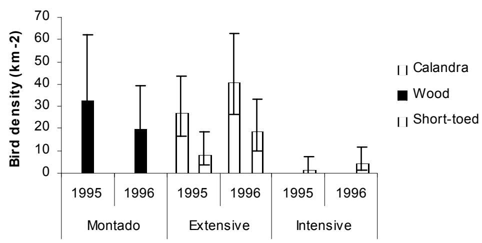


Fig 1. Densities (number km-2 with 95% confidence limits) of Calandra Lark, Woodlark and Short-toed Lark in April in relation to three Alentejo farming systems.

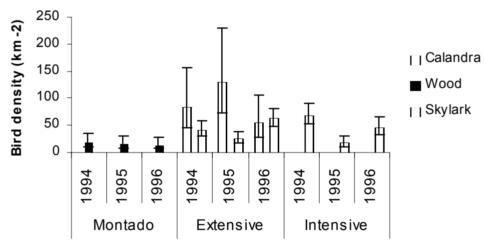


Fig 2. Densities (number km-2 within 95% confidence limits) of Calandra Lark, Woodlark and Skylark in December in relation to three Alentejo farming systems.

totalling 155 000 ha. Within this region, three land-use systems were recognised: intensive agriculture, extensive agriculture and *Montado* (Tab. 1).

The intensive agriculture category is characterised by a greater frequency (>55%) of heavy soils, much of the area being irrigated. Wheat *Triticum aestivum* and barley *Hordeum distichum* are the main cereal crops and silage grass *Lolium sp*, sunflower *Helianthus annuus*, sugar beet *Beta vulgaris* and oilseed rape *Brassica napus* are also grown. Wheat yields are 2.5-3.5 tonnes $ha^{-1}$ without irrigation but can be almost doubled with full irrigation (P. Eden *pers comm* 1998). There are short rotations with little or no fallow (*e.g.* sunflower / 1st cereal / 2nd cereal). This system requires frequent use of fertiliser (130 units $N_2$ $ha^{-1}$ (P. Eden *pers comm* 1998)) and herbicides, relative to the other land-use categories. With the exception of some olive *Olea europaea* groves, there is little tree cover.

The extensive agriculture category is characterised by thin soils and by the largest average farm size of the three categories (Tab. 1). There is no irrigation and the fallow area is relatively high. A typical rotation takes the form: plough fallow / 1st cereal / 2nd cereal / fallow / fallow. Fallow periods often last five years or more (Rio Carvalho *et al.* 1995). Wheat yields are 1.5-2.5 tonnes $ha^{-1}$, yields at the lower end of this range being more common (P. Eden *pers comm* 1998). Triticale *Triticum aestivum x Secale cereale* and oats *Avena sativa* are frequently grown in the extensive category and grazed or cut for silage. The incorporation of a fallow period into the rotation, and the relatively low potential yields are associated with considerably lower annual inputs than in the intensive category.

*Montado* (equivalent to the Spanish *Dehesa*) is characterised by thin soils and tree cover, dominated by holm oak *Quercus rotundifolia* and cork oak *Q. suber* at up to 20 trees $ha^{-1}$ (mean=10.5 trees $ha^{-1}$, se=0.7). Like the extensive category, there is no irrigation and the fallow area is high. A typical rotation is similar to that of the extensive category, although the fallow stage is often longer and forage lupins *Lupinus luteus* may be included.

Sheep *Ovis aries*, cattle *Bos taurus* and pigs *Sus scrofa* are kept in all three land-use categories. Zero grazing is adopted on some farms in the intensive category but livestock normally graze fallows. Tab. 1 lists the proportion of crops and livestock in each category.

## 3. Methods

A total of 115 250 m transect counts, starting at 1 km grid intersections and stratified by land-use categories, were conducted along a random bearing (*Montado* n=42, Extensive n=42, Intensive n=31). Transect counts were conducted by a single observer in the first three hours after dawn, or the two hours before dusk in December (1994, 1995 and 1996) and April (1995 and 1996). Perpendicular distances from the transect line to each detected bird were estimated visually to the nearest 10 m. Detectability was assumed to be the same in each of the three land-use categories, even in *Montado*, because habitat structure is relatively open. The number of birds seen together at an observation was recorded.

Density estimates were calculated for each land-use category using line transect sampling and the computer program DISTANCE (Buckland *et al.* 1993, Laake *et al.* 1993). Analyses used clusters (<1 individual) as analytical units, and untruncated perpendicular distance data. A variety of recommended robust estimators implemented by DISTANCE was used, the final model selected in each case being the one with the lowest Akaike's Information Criterion value (Buckland *et al.* 1993). Differences in bird abundance between land-use categories were tested using two-way ANOVA (year×land-use) and LSD post-hoc tests (at P<0.05) log-transformed data.

## 4. Results

Densities of the four species are presented at Figs 1 (April) & 2 (December). Woodlark occurred only in *Montado* and Calandra Lark only in extensive farmland, in both April and December. Differences between land-use categories were significant for Woodlark in December ($F_{2,5}$=210.4, P<0.001) but not in April, and for Calandra Lark in both December ($F_{2,5}$=15.0, P<0.01) and April ($F_{2,2}$=25.2, P<0.04). Short-toed Lark occurred in both extensive and intensive farming systems, with a higher density in the former, although this difference was not significant. Skylark was absent in April and occurred at similar densities in both extensive and intensive systems in December.

## 5. Discussion

The international distribution of Woodlark is confined almost exclusively to Europe, between 12% and 34% of this population occurring in Portugal (Tucker & Heath 1994). This species occurs in several Portuguese forest systems, but within farming systems, the species is restricted to *Montado*. Large areas of *Montado* have been cleared since the 1930s, and continued local clearance of trees, such as that associated with the development of the Alqueva reservoir, is likely to have severe consequences for Woodlark and other species associated with *Montado* in Alentejo (Vieira & Eden 1995). Changes in scrub clearance frequency and practices, grazing densities and livestock types also influence tree health and habitat structure (Pinto-Correia & Mascarenhas 1999). Financial support is currently available for the re-establishment of *Montado* within Alentejo under EU Reg. 2080/92 so that the total area of *Montado* is currently not experiencing substantial change.

However, this is a long-term project, and the habitat is also threatened by the spread of a fungal disease that is causing widespread destruction of oak trees across the Iberian Peninsula (Brasier 1992).

Large- and medium-sized invertebrates such as grasshoppers (Orthoptera), spiders (Araneae) and caterpillars (Lepidoptera and Symphyta larvae) form an important nestling food source for each of the species considered here (Cramp 1988). Low pesticide inputs and continued incorporation of fallow periods into arable rotations are associated with higher densities of these invertebrates than in intensively managed agricultural systems in Alentejo (Stoate *et al.* 2000). This might explain the low breeding abundance of larks in intensively managed systems, for as Ewald and Aebischer (1999) have shown, there is a relationship between pesticide use, invertebrate abundance and breeding abundance of Grey Partridge *Perdix perdix* and Corn Bunting *Miliaria calandra* in the U.K. Like larks, Corn Bunting is also present at relatively low densities in intensively managed agricultural systems in Alentejo (Stoate *et al.* 2000).

Skylark is absent from all farming systems during April, confirming the breeding distribution of Rufino (1989). However, in December this species occurs in both extensive and intensive landscape categories. This may be explained by the species using cereal crop leaves as a winter food source, although Calandra Lark, which also feeds on cereal leaves (Cramp 1988), is absent from intensively managed systems. About one third of the European Calandra Lark population occurs in the Iberian Peninsula where the species is declining in numbers (Tucker & Heath, 1994). Moreira (1999), working in the same area, found that Calandra Lark was strongly associated with low vegetation on fallows within extensive farming systems. However, the ecological requirements of Calandra Lark in Portugal throughout the year are poorly understood and require further investigation if appropriate management is to be adopted for this species.

Short-toed Lark occurred in intensive agricultural systems, but was present at low densities and generally was associated with non-crop habitats such as tracks and other uncultivated areas. Moreira (1999) reported an association between Short-toed Lark abundance and bare ground within fallows. Like Calandra Lark, this species appears to benefit from the maintenance of extensive farming systems. This habitat supports more bird species of current conservation concern than does any other (Tucker & Heath 1994, Santos 1996, Araújo *et al.* 1996). Some of these species (*e.g.* Lesser Kestrel *Falco naumanni*, Great Bustard *Otis tarda*, Little Bustard *Tetrax tetrax* and Roller *Coracias garrulus*) are highly valued by birdwatchers, and a small-scale tourist industry is currently developing around the presence of these species on extensively managed farmland in Portugal.

Farm tourism is also developing in Spain (Garcia-Ramon *et al.* 1995) and provides an opportunity for alternative income for management of extensive farming systems, either directly through accommodation on farms, or indirectly, as in Alentejo, through EU Rural Development support for maintaining such systems. However, many steppe species are susceptible to disturbance and access requires careful management. This does not preclude the potential for combining public and private support for extensive farming systems that support larks and other species associated with extensively managed arable landscapes.

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# Use of Playback in Estimating Numbers of Bearded Tit *Panurus biarmicus* Outside the Breeding Season.

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## 1. Introduction

Although the European population of Bearded Tit *Panurus biarmicus* is largely sedentary, there have been only a few studies of Bearded Tits outside the breeding season. The camouflaging effect of their habitat (reedbeds) and their reduced vocal activity are the main difficulties presented during this time (Wawrzyniak and Sohns 1986). Nevertheless, studies outside the breeding season may be crucial for monitoring changes in the numbers and range of the Bearded Tit population (Gosler & Mogyorósi 1997). In this study of Bearded Tits outside the breeding season, the efficiency of undertaking point counts that were combined with playback was assessed in comparison with traditional methods.

## 2. Study area and methods

The survey took place in Western Poland in landscape dominated by arable fields. In a 40 km×52 km plot, I selected randomly 45 reed-beds whose width was at least 5m. At each reedbed I used a fixed observation post at the edge of the reeds. I controlled each site 10 times between the end of September 2000 and the beginning of March 2001 at 10-15 days intervals, performing the surveys between 0700 and 1500 during fine weather. On each occasion I censused Bearded Tits using three different variants of point count, applied in the following sequence:

1. 180 seconds of observations without any stimulation.
2. 140 seconds of observations preceded by 40 seconds of noisy walking through reeds.
3. 140 seconds of observations preceded by 40 seconds of playback of taped contact calls. (6 W).

If Bearded Tits were present during census, I played up to 300 seconds of playback calls to attract birds as close as possible to the observer. I measured the time from the beginning of playback to the first visual observation of the birds. When the birds approached, I recorded the following information: number of birds, sex, the nearest approach distance to the observer (to the nearest half metre) and type of behaviour. I used SPSS software for the statistical analyses.

## 3. Results

During the study, I recorded Bearded Tits at 29 sites (64.4% of controlled sites). I recorded 348 birds from 112 visits. The

Tab. 1. Results of census obtained by point counts without stimulation (W), point counts without stimulation but preceded by noisy walking through reeds (W+N) and playback technique (P).

| | W | W+N | P |
|---|---|---|---|
| number of sites with at least one record | 20 | 24 | 27 |
| number of records | 49 | 76 | 109 |
| number of individuals | 141 | 180 | 297 |
| males [%] | 11 | 11 | 35 |
| females [%] | 9 | 8 | 28 |
| unidetified idividuals [%] | 80 | 81 | 37 |

overall results obtained from the three census methods are at Tab. 1.

The efficiency of playback in detecting Bearded Tits was independent of the date of the census (differences between September-December and January-March were tested, $\chi^2$=0.50, DF=1, p=0.48, N=33) and the number of individuals at each site (differences between flocks detected through the use of different censusing methods were tested using Kruskal-Wallis ANOVA; H=2.28, p<0.33, N=112). Bearded Tits were attracted by playback successfully in 71 cases (64%). In 24 cases (21%) there were only vocal responses. The efficiency of attracting the species was independent of the method of detection used ($\chi^2$=0.93, DF=2, p=0.63, N=71) and the frequency of playback use at a particular site (differences between sites with 1-4 and 4-8 records were tested, $\chi^2$=0.51, DF=1, p=0.47, N=71). The mean time of attracting birds was 93.17±81.21 seconds. The mean time did not vary in relation to the number of individuals at a site (differences between sites with 1-2 and 3-8 individuals were tested using Mann-Whitney U test, z=-0,37, p=0.71, N=70,). The mean closest distance to which birds approached was 3.00±1.84 m (min=0.5 m, max=15 m, N=112), there being no difference between males and females (Mann-Whitney U test, z=-0,47, p=0.64, N=118).

The behaviour of birds attracted by playback could be classed as 'natural' or 'nervous'. The former, typical of *c*80% of birds, was characterised by occasional calls, pluming perching or foraging. The latter, applicable to the remainder of the birds that called frequently, moved restlessly between reed stems, sought the playback source intensively and rarely foraged. A total of 136 of the observed birds foraged.

## 4. Discussion

The results obtained suggest that the occurrence of Bearded Tit outside the breeding season might be quite widespread and regular even when there are no irruptions (compare Spitzer 1972). Accidental information on the numbers of Bearded Tit outside the breeding season, that is the main source for regional monographs (*e.g.* Kuźniak 2000), may have been an underestimate, by a factor of two (Tab. 1). It is therefore likely that distrib-

Tab. 2. Spearman rank correlation matrix for the number of records and individuals recorded, comparing the two non-playback census methods with the playback method (n=10). (See Tab. 1 for explanation of the categories).

| Census Method With Playback | W | W+N |
|---|---|---|
| $P_{records}$ | r=0.59, p=0.073 | r=0.72, p=0.018 |
| $P_{number\ of\ individuals}$ | r=0.69, p=0.028 | r=0.82, p=0.004 |

ution and habitat selection analyses based on those data are in error, and may be even less reliable because of the low delectability of the species (see Tab. 2). In theory, more precise methods of census based on detailed counts along reedbed edges (von Einstein 1985) or on mist-netting (von Dürr *et al.*) may given more accurate results than the methods I have presented of point count variants. In practice, however, these precision methods are expensive of time and effort, and so regular surveys usually are limited to one site (*e.g.* a lakeside) or require several years to gather sufficient data. It is very probable that playback increases count efficiency significantly and improves results of mist-netting and behavioural observations. With my method, the majority of birds would approach very closely, showing no fear of the observer and behaving naturally. Another advantage of the playback method is its equal efficiency with respect season, flock size, frequency of use or the presence of noise caused by walking through the reeds. Results presented below are undoubtedly preliminary, and further studies are needed, especially for different populations and in other habitats and during different weather conditions.

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