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Impact of industrial emissions on bird populations breeding in mountain spruce forests in Central Europe

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The spruce forest complex in the Krkonoše Mts. of Czechoslovakia is among those most affected by industrial emissions in Europe. Breeding birds in this area have been censused using line transects (total 179 km) and point counts (484 points) since 1983 in four different habitats from almost healthy beech-spruce and spruce forests to heavily damaged and dead spruce forests and subsequent clearcuts.

The total bird density was highest in good quality stands (319–430 pairs/km^2 or 100% in 1984–87) and gradually decreased with increasing damage to the forest (80–89% in slightly and moderately damaged, and 50–55% in heavily damaged and dead forests). The dominant species exhibited a similar decrease (especially *Regulus, Turdus, Troglodytes, Fringilla* and *Parus*).

In 1984–87 the total density of breeding birds decreased by about one-third in the different habitats. Out of the 15 most abundant species the densities decreased significantly in 6 (40%), increased in 1 and fluctuated or remained constant in 8. Such decreases were not typical of larger areas and it is therefore argued that the trends can be attributed to the effect of industrial emissions in the study area.

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1. Introduction

In recent years the influence of industrial emissions on bird populations has been observed with growing intensity, mostly in Central Europe. This fact is related to the extensive decline of forest stands, which are one of the main factors determining the structure and abundance of forest bird communities. The direct use of birds for bioindication purposes is limited (cf. Morrison 1986), but long-term monitoring of breeding bird populations can provide, together with other data, valuable information on the mechanism of changes taking place in the environment.

Several papers have dealt with the impact of emissions on birds only locally by examining morphological changes of individual birds or reactions of the bird communities near the centre of pollution (Kempf & Sittler 1977, Feriancová & Kalivodová 1965, Pinowska et al. 1981, Slobodník 1981 and others). More recent studies have approached this problem on a larger geographical scale (Kux 1982, Šťastný & Bejček 1983, Hölzinger & Kroymann 1984, Pelc 1986, Šťastný et al. 1987).

Thanks to their relatively small area and to a diverse topography with a number of vegetation belts, the Krkonoše Mts. are an ideal area for monitoring important components of the natural environment ranging from submontane to alpine belts. Their unique geographical location and geomorphology have created optimal conditions for plant and animal communities, where alpine and arctic, lowland and mountain elements have mingled. The location of the Krkonoše Mts. (a prominent frontier of the Central European mountains) and their anemo-orographic systems (complex of ecological factors with a strong influence on the local nature — cf. Jeník 1961) are responsible for the present natural richness of this area. In the last 10–15 years, however, both the factors have shown their negative aspects as well. The Krkonoše Mts. are located in the vicinity (50–120 km) of important industrial centres in Poland, the GDR and NW Czechoslovakia. Industrial emissions, brought by prevailing western air currents, adversely affect the quality of forest stands, soil and surface waters, not only on windward slopes and mountain ridges but also in leeward mountain valleys and corries. As a result dominant spruce forests are rapidly dying out on a large scale (*Picea abies* forms 86% of current forests) and also beech stands in the foot-hills are weakened. For these reasons the Krkonoše National Park (385 km², about 84% forests) has been ranked among the 11 most endangered national parks of the world (IUCN 1984).

In order to monitor changes in the natural environment of the Krkonoše Mts., permanent research plots were established in 1983. The present paper is a result of the first five years of monitoring bird populations in the environment strongly affected by the complex of factors connected with "acid rain". Although not all of the data have been analysed until now, the results give a warning. They support the arguments of international movements for decreasing industrial emissions.

2. Material and methods

Breeding bird populations have been monitored in the Krkonoše Mts. (= the Giant Mts., NE Bohemia, Czechoslovakia, about 50°40'N, 15°40'E) since 1983. Because of the extensive area (385 km²) and a varied mosaic of habitats relative census methods were preferred, because they make it possible to survey a large area and provide satisfactory results for monitoring purposes (cf. Bibby & Robins 1985).

Line transects: Birds were censused on transects with a length of 2.2–8.5 km (total 178.4 km in 1983–87) and a width of 100 m. The transect width was chosed with regard to the favourable structure of the habitats studied (shrub layer not marked, often missing, made it possible to count birds up to 50 m on each side of the transect) (Tomialojć 1981, Verner 1985).

Point counts: A modified I.P.A. method (Blondel et al. 1970) was used. Birds were counted for 5 minutes (cf. Janda 1982) within a circle of radius 100 m, making a break for the point count during the line transect. All points (total 484 in 1983–87) were thus placed along most transects (the distance between points was 200–400 m).

Birds on the same transects and points were counted twice during the breeding season (second half of May, second half of June), early in the morning (4.00–9.00 CET), in satisfactory weather conditions. The higher abundance value of each species from the two counts was used. The transect routes and counting points were identical in all study years.

The acquired data of both the methods were converted to the number of pairs according to commonly used criteria: singing male, pair, family, occupied nest, feeding bird = 1 pair, individually seen or heard bird = 0.5 pair (Blondel et al. 1970, Ferry 1974).

The density (D_i, in pairs/km² of transects resp. counting points) of species i was used in calculating the dominance (d_i), giving the proportion (in %) of species i in the community. The dominance values can be used to calculate a number of well-known measures related to the shape of the species-abundance distribution of the community, such as index of diversity (Shannon & Weaver 1949), evenness (Sheldon 1969), index of dominance concentration (Simpson 1949), and dominance index (McNaughton 1967). As all of these indices tend to be correlated fairly strongly, Shannon's index (bird species diversity BSD, 2-based logarithms) will be used in this paper. However, the calculations were also made using the other indices, but the main biological conclusions were unaltered.

The results of point counts (densities, number of species per point) were expressed as the average values and complemented by the standard deviation (*SD*). The z-test (Fowler & Cohen 1986) was used for testing differences between means. The trends, both in time and forest damage succession, were tested using the regression analysis (Anděl 1985, Fowler & Cohen 1986). If a rectilinear relation was not confirmed (test of linearity), the logarithmic transformation of a curvilinear relationship was used ($\ln y = bx + a$). The significance of the regressions was determined by the one-tailed t-test. The chi-square statistic was calculated to find out possible differences in number of species.

Line transects and counting points covered the basic habitats over the entire area of the national park, from foot-hills to the alpine timberline. The habitats which were monitored were grouped according to the following criteria (Kučera 1978):

BS (84 counting points): almost healthy beech-spruce (80–90% *Picea abies*) and spruce forests at 580–1030 m a.s.l., proportion of damaged trees up to 5%, most trees with 6 and more years of needles, canopy unchanged).

SD (128 points): slightly damaged spruce stands at 670–1030 m a.s.l., proportion of damaged trees 6–30%, trees with 3–5 years of needles.

MD (87 points): moderately damaged spruce stands at 880–1070 m a.s.l., proportion of damaged trees 31–50%, trees with 2–5 years of needles, all tree crowns partly dry.

HD (112 points): heavily damaged and dead spruce stands at 920–1230 m a.s.l., 51–100% heavily damaged trees with 0–3 years of needles, majority of trees dying.

All the forests studied were, with minor exceptions (natural stands), about 60–140 year-old plantations (under 60 years: 19%, 61–80: 19%, 81–100: 21%, 101–120: 17%, above 121 years: 23%), with a minimal (or absent) shrub layer. The herb layer (especially *Calamagrostis villosa* and *Avenella flexuosa*) increased with increasing amount of damage and forest thinning.

CC (73 points): clearings after felling heavily damaged forests, with islands of dead spruce stands left (up to 5 ha) at 960–1300 m a.s.l., herb layer with 5–10 year-old *Picea abies* plantations, numerous stumps, windfalls and heaps of branches (slash).

Transects could not always be located in one habitat only. The habitats were therefore divided into 3 groups: BS (38.5 km transects), SMD (SD+MD, 36.8 km) and HDC (HD followed by clearcutting, 103.1 km).

3. Results

Breeding bird density was the highest in healthy forests and decreased gradually with increasing damage to spruce stands (Tables 1–2).

The results of the line transects indicated a total of 53 species at a density of 319–430 pairs/km² (passer-

Table 1. Line transects: total densities (pairs/km²), bird species diversity and number of species in variously damaged spruce forests in 1984–1987 (for BS, SMD, HDC see Table 3).

	1984	1985	1986	1987
Total density				
BS*C	430.6	362.8	337.2	318.9
SMD**C	379.9	324.5	282.1	257.1
HDC*C	238.5	187.6	180.3	160.9
Bird species diversity				
BS*L	3.69	3.73	3.87	4.18
SMD^NS	3.80	3.36	3.71	3.86
HDC^NS	3.76	3.42	3.88	3.88
Number of species				
BS (53 sp.)^NS	42	39	38	41
SMD (52)^NS	43	37	38	38
HDC (55)^NS	44	33	46	45

Trends: C – curvilinear, L – linear; * $P<0.05$, ** $P<0.01$, NS – not significant.

ines: 45 spp., D=312–425 pairs/km²) in minimally affected beech-spruce and spruce forests (BS). The density in slightly to moderately damaged spruce stands reached 80.4–89.4% D_{BS} with 52 noted species (passerines: 44 spp., D=253–375 pairs/km²) and decreased to 50.4–55.4% D_{BS} for 55 species (passerines: 47 spp., D=159–235 pairs/km²) in heavily damaged and dead forests followed by clearcutting (Table 1).

Point counts confirmed the tendencies outlined by the results of the line transects (Table 2): the decrease of total density along the forest damage succession from healthy (D=356.0 pairs/km²) to damaged and dead stands (D=213.4 pairs/km²). In clearcuts the density increased roughly to the level of moderately damaged forests.

In regard to the density of the dominant and influent species (d>2%) in the study habitats, different reactions were exhibited by individual species (Fig. 1, Tables 3–4). The density of a few species decreased significantly with the increasing amount of damage from the healthy forests to the clearcuts (*Regulus* spp., *Turdus* spp., *Parus ater, Troglodytes troglodytes, Fringilla coelebs* etc.). The density of other species remained about the same in all habitats (*Prunella modularis, Erithacus rubecula, Phylloscopus trochilus*), or even increased with the increasing forest damage (*Anthus* spp., *Motacilla cinerea, Phoenicurus ochruros,* partly also *Phoenicurus phoenicurus* in BS to MD).

The results of the line transects did not show any difference in bird species diversity in various habitats (Table 1). In contrast to the healthy stands (significant increase of BSD in 1984–87), no trends were noted in BSD in damaged forests during the study period. The point counts, however, registered a significant decrease of BSD along the forest damage succession (BS-HD, Table 2) and an increase of BSD in the clearcuts compared to healthy or slightly damaged stands.

No difference was found in the total number of species observed in the habitats studied (Tables 1–2), but the average number of species per point in point counts (Table 2) decreased significantly from the healthy (8.6 species/point) to the heavily damaged forests (4.9 species/point) and then increased again in clearings.

In 1983(84)–1987 the total density of breeding birds in individual transects decreased to 49–91% of the initial density in 1983 (or 1984). The decrease in density in better quality stands was more moderate (–26%) than in damaged spruce forests (about –32% in both habitats) (Table 1); the differences in slopes of these trends, however, are not significant.

Out of 15 species whose average dominance exceeded 2% in at least one of the habitats investigated (Table 5), the density decreased significantly in 6 species (40%, gradual decrease especially in *Regulus*

Table 2. Point counts: total densities (pairs/km² ± SD), average number of species/point ± SD, total number of species, and bird species diversity in variously damaged spruce forests (for BS, SD, MD, HD, CC see Table 4).

	BS	SD	MD	HD	CC
Total density**C	356.0±100.3	302.1±90.1	253.6±76.7	213.4±85.6	261.4±77.3
Average no. of species**L	8.6±2.9	7.3±2.0	6.0±1.9	4.9±2.0	6.5±1.9
Total no. of species^NS	48	51	38	36	52
BSD**L	4.14	3.96	3.68	3.52	4.04

Trends: C – curvilinear, L – linear (relationships without CC tested); **$P<0.01$. Differences in total densities and average number of species were highly significant ($P<0.01$) for all pairs of habitats, except MD-CC ($P>0.1$).

Table 3. Line transects: average densities (pairs/km^2) and dominance (%, in parentheses) of bird species breeding in almost healthy beech-spruce and spruce forests (BS, 36 km transects), slightly and moderately damaged spruce forests (SMD, 36.8 km), and heavily damaged and dead spruce forests followed by clearcutting (HDC, 93.6 km) in 1984–1987.

	BS	SMD	HDC		BS	SMD	HDC
Accipiter nisus	0.3 (0.1)	–	–	*S. atricapilla*	27.5 (7.6)	15.2 (4.9)	3.6 (1.9)
Buteo buteo	0.6 (0.2)	0.3 (0.1)	0.2 (0.1)	*Phylloscopus bonelli*	–	–	0.1 (0.1)
Falco tinnunculus	–	–	0.2 (0.1)	*P. sibilatrix*	3.9 (1.1)	1.9 (0.6)	0.1 (0.1)
Lyrurus tetrix	–	–	0.2 (0.1)	*P. collybita*	13.9 (3.8)	8.7 (2.8)	6.2 (3.2)
Columba oenas	–	0.1 (0.0)	–	*P. trochilus*	10.6 (2.9)	8.7 (2.8)	4.5 (2.3)
C. palumbus	1.5 (0.4)	1.2 (0.4)	0.6 (0.3)	*Regulus regulus*	20.7 (5.7)	15.5 (5.0)	7.4 (3.8)
Streptopelia turtur	0.4 (0.1)	0.3 (0.1)	–	*R. ignicapillus*	7.5 (2.1)	6.5 (2.1)	1.2 (0.6)
Cuculus canorus	0.7 (0.2)	1.5 (0.5)	1.0 (0.5)	*Muscicapa striata*	0.3 (0.1)	–	–
Jynx torquilla	–	0.3 (0.1)	0.1 (0.1)	*Ficedula parva*	–	0.8 (0.3)	–
Picus canus	0.4 (0.1)	–	–	*F. hypoleuca*	2.5 (0.7)	1.2 (0.4)	0.6 (0.3)
Dryocopus martius	0.4 (0.1)	0.1 (0.0)	0.1 (0.0)	*Pařus palustris*	0.6 (0.2)	–	–
Dendrocopos major	0.8 (0.2)	0.1 (0.0)	0.3 (0.2)	*P. montanus*	0.4 (0.1)	–	0.1 (0.1)
Lullula arborea	–	–	0.1 (0.1)	*P. cristatus*	2.2 (0.6)	1.0 (0.3)	1.0 (0.5)
Alauda arvensis	2.5 (0.7)	0.5 (0.2)	–	*P. ater*	9.6 (2.6)	9.8 (3.1)	4.6 (2.4)
Hirundo rustica	0.4 (0.1)	0.3 (0.1)	0.1 (0.0)	*P. caeruleus*	0.7 (0.2)	0.3 (0.1)	–
Anthus trivialis	14.9 (4.1)	19.0 (6.1)	18.8 (9.8)	*P. major*	5.1 (1.4)	2.9 (0.9)	0.9 (0.4)
A. pratensis	–	1.1 (0.3)	4.5 (2.3)	*Sitta europaea*	1.3 (0.3)	–	–
A. spinoletta	–	–	0.1 (0.0)	*Certhia familiaris*	2.9 (0.8)	3.7 (1.2)	1.4 (0.7)
Motacilla cinerea	5.3 (1.5)	5.3 (1.7)	7.6 (4.0)	*Lanius collurio*	–	–	0.1 (0.1)
M. alba	0.1 (0.0)	0.4 (0.1)	0.3 (0.2)	*Garrulus glandarius*	1.8 (0.5)	1.4 (0.4)	0.3 (0.2)
Cinclus cinclus	0.7 (0.2)	0.5 (0.2)	0.1 (0.1)	*Nucifraga caryocatactes*	–	0.1 (0.0)	0.1 (0.1)
Troglodytes troglodytes	8.1 (2.2)	7.2 (2.3)	3.6 (1.9)	*Corvus corone*	0.1 (0.0)	0.4 (0.1)	0.4 (0.2)
Prunella modularis	17.8 (4.9)	17.1 (5.5)	14.2 (7.4)	*C. corax*	–	0.1 (0.0)	–
Erithacus rubecula	18.9 (5.2)	19.2 (6.2)	10.7 (5.6)	*Sturnus vulgaris*	0.1 (0.0)	0.7 (0.2)	–
Phoenicurus ochruros	0.3 (0.1)	1.6 (0.5)	4.4 (2.3)	*Fringilla coelebs*	128.3 (35.4)	118.8 (38.2)	69.8 (36.4)
P. phoenicurus	4.2 (1.1)	2.2 (0.7)	2.1 (1.1)	*Serinus serinus*	0.3 (0.1)	–	–
Saxicola rubetra	0.3 (0.1)	0.3 (0.1)	0.5 (0.3)	*Carduelis spinus*	2.4 (0.7)	2.7 (0.9)	1.8 (0.9)
Turdus torquatus	–	0.4 (0.1)	1.7 (0.9)	*C. flammea*	1.3 (0.3)	1.5 (0.5)	3.3 (1.7)
T. merula	6.1 (1.7)	4.1 (1.3)	0.4 (0.2)	*Loxia curvirostra*	1.3 (0.3)	2.7 (0.9)	1.3 (0.7)
T. pilaris	–	–	0.3 (0.2)	*Carpodacus erythrinus*	–	–	0.1 (0.1)
T. philomelos	16.9 (4.7)	11.7 (3.8)	5.0 (2.6)	*Pyrrhula pyrrhula*	3.6 (1.0)	1.0 (0.3)	1.1 (0.6)
T. viscivorus	1.7 (0.5)	4.2 (1.4)	2.7 (1.4)	*Coccoth. coccothraustes*	0.3 (0.1)	0.4 (0.1)	0.1 (0.1)
Sylvia curruca	0.8 (0.2)	–	0.4 (0.2)	*Emberiza citrinella*	3.5 (1.0)	1.6 (0.5)	1.0 (0.5)
S. communis	2.8 (0.8)	2.2 (0.7)	0.4 (0.2)				
S. borin	3.1 (0.8)	2.2 (0.7)	0.2 (0.1)	Total no. of pairs	1305	1144	1795

spp., *Fringilla coelebs* and *Erithacus rubecula*), increased in 1 species (*Anthus trivialis*) and fluctuated or remained constant in 8 species (53%) during the study period (Fig. 1).

4. Discussion

The present study reveals an important negative relationship between the amount of forest damage and breeding bird density and bird species diversity. By contrast, both these characteristics tend to increase in newly formed clearings.

No significant decrease was observed in bird species diversity based on the line transect results along the forest damage succession. It is a well known phenomenon that an intermediate disturbance can increase the species diversity (Connell 1978), and the noted increase in BSD in healthy forests in 1984–87 may be explained by this hypothesis. The reaction, however, is a result of long-term evolutionary adaptation and does not apply to the effect of industrial emissions, an example of a drastic and rapid disturbance due to human activities (cf. Connell 1978). It seems that the methodological aspect can provide a better explanation. In the case of line transects, the

Table 4. Point counts: average densities (pairs/km^2) and dominances (%, in parentheses) of bird species breeding in almost healthy beech-spruce and spruce forests (BS, 84 points), in slightly (SD, 128), moderately (MD, 87) and heavily damaged and dead (HD, 112) spruce forests, and in clearings (CC, 73).

	BS	SD	MD	HD	CC
Anas platyrhynchos	–	0.2 (0.1)	–	–	–
Accipiter nisus	–	0.1 (0.0)	–	–	–
Buteo buteo	0.4 (0.1)	0.6 (0.2)	–	0.3 (0.1)	1.1 (0.4)
Falco tinnunculus	–	–	–	–	0.4 (0.2)
Tetrao urogallus	–	0.1 (0.0)	–	–	–
Lyrurus tetrix	–	–	–	0.1 (0.1)	0.4 (0.2)
Columba palumbus	3.4 (1.0)	1.2 (0.4)	1.3 (0.5)	–	0.2 (0.1)
Streptopelia turtur	0.4 (0.1)	0.1 (0.0)	–	–	–
Cuculus canorus	0.8 (0.2)	1.0 (0.3)	1.3 (0.5)	1.0 (0.5)	2.4 (0.9)
Jynx torquilla	–	–	–	–	0.9 (0.3)
Picus canus	0.2 (0.1)	–	–		–
Dryocopus martius	0.4 (0.1)	0.1 (0.0)	0.2 (0.1)	0.3 (0.1)	0.4 (0.2)
Dendrocopos major	1.7 (0.5)	0.1 (0.0)	–	–	0.9 (0.3)
Lullula arborea	–	–	–	–	0.4 (0.2)
Alauda arvensis	2.7 (0.7)	0.5 (0.2)	–	–	0.9 (0.3)
Hirundo rustica	0.2 (0.1)	–	–	0.1 (0.1)	1.3 (0.5)
Delichon urbica	–	–	–	–	0.2 (0.1)
Anthus trivialis^{*C}	10.4 (2.9)	17.3 (5.7)	22.5 (8.9)	16.9 (7.9)	51.0 (19.5)
A. pratensis^{*C}	–	–	0.4 (0.1)	0.6 (0.3)	17.4 (6.7)
Motacilla cinerea^{*L}	–	5.5 (1.8)	7.5 (3.0)	8.4 (3.9)	9.2 (3.5)
M. alba	0.2 (0.1)	0.1 (0.0)	–	–	0.9 (0.3)
Cinclus cinclus	–	0.1 (0.0)	0.2 (0.1)	–	0.2 (0.1)
Troglodytes troglodytes^{**C}	9.1 (2.6)	7.7 (2.6)	5.3 (2.1)	4.4 (2.1)	4.4 (1.7)
Prunella modularis^{NS}	18.0 (5.1)	20.0 (6.6)	15.0 (5.9)	15.9 (7.5)	26.6 (10.2)
Erithacus rubecula^{NS}	19.7 (5.5)	22.1 (7.3)	18.3 (7.2)	18.5 (8.7)	9.2 (3.5)
Phoenicurus ochruros^{**C}	0.6 (0.2)	0.7 (0.2)	2.6 (1.0)	3.4 (1.6)	8.5 (3.3)
P. phoenicurus^{NS}	0.4 (0.1)	3.2 (1.1)	7.0 (2.7)	1.7 (0.8)	2.6 (1.0)
Saxicola rubetra	–	–	–	–	2.0 (0.8)
Turdus torquatus	–	0.2 (0.1)	0.4 (0.1)	2.4 (1.1)	0.7 (0.3)
T. merula^{*L}	9.7 (2.7)	5.3 (1.8)	0.7 (0.3)	–	1.3 (0.5)
T. pilaris	–	0.1 (0.0)	–	0.1 (0.1)	0.4 (0.2)
T. philomelos^{**L}	18.0 (5.1)	16.5 (5.5)	8.8 (3.5)	9.0 (4.2)	4.6 (1.8)
T. viscivorus	4.4 (1.2)	5.7 (1.9)	4.9 (1.9)	3.6 (1.7)	5.0 (1.9)
Sylvia curruca	1.3 (0.4)	1.2 (0.4)	0.4 (0.1)	–	0.4 (0.2)
S. communis	3.0 (0.9)	0.2 (0.1)	0.7 (0.3)	–	0.7 (0.3)
S. borin	3.4 (1.0)	2.2 (0.7)	–	–	–
S. atricapilla^{*C}	24.3 (6.8)	21.9 (7.2)	5.5 (2.2)	6.0 (2.8)	3.9 (1.5)
Phylloscopus bonelli	–	–	–	0.3 (0.1)	–
P. sibilatrix	6.4 (1.8)	2.2 (0.7)	0.4 (0.1)	–	–
P. collybita^{*C}	16.1 (4.5)	17.7 (5.8)	8.8 (3.5)	7.7 (3.6)	8.7 (3.3)
P. trochilus^{NS}	9.9 (2.8)	10.9 (3.6)	1.8 (0.7)	3.0 (1.4)	8.7 (3.3)
Regulus regulus^{**C}	20.1 (5.6)	15.2 (5.0)	16.1 (6.3)	9.7 (4.5)	6.1 (2.3)
R. ignicapillus^{**C}	10.2 (2.9)	4.0 (1.3)	1.5 (0.6)	1.8 (0.9)	0.4 (0.2)
Ficedula hypoleuca	0.8 (0.2)	1.2 (0.4)	1.6 (0.6)	–	–
Parus palustris	0.2 (0.1)	0.2 (0.1)	–	–	–
P. montanus	0.8 (0.2)	0.1 (0.0)	–	0.3 (0.1)	–
P. cristatus	5.3 (1.5)	1.9 (0.6)	2.7 (1.1)	1.8 (0.9)	0.4 (0.2)
P. ater^{**L}	15.0 (4.2)	10.3 (3.4)	11.5 (4.5)	4.0 (1.9)	1.7 (0.7)
P. caeruleus	1.5 (0.4)	–	0.2 (0.1)	–	–
P. major	2.8 (0.8)	3.2 (1.1)	–	0.3 (0.1)	0.4 (0.2)
Sitta europaea	1.5 (0.4)	0.1 (0.0)	–	–	–
Certhia familiaris	5.9 (1.6)	1.4 (0.5)	3.1 (1.2)	1.1 (0.5)	0.4 (0.2)
Lanius collurio	–	–	–	–	0.7 (0.3)
Garrulus glandarius	1.7 (0.5)	0.7 (0.2)	0.4 (0.1)	0.1 (0.1)	0.2 (0.1)
Nucifraga caryocatactes	0.2 (0.1)	0.2 (0.1)	0.4 (0.1)	0.1 (0.1)	–
Corvus corone	0.4 (0.1)	0.7 (0.2)	0.2 (0.1)	0.1 (0.1)	0.7 (0.3)
C. corax	–	–	–	–	0.4 (0.2)
Sturnus vulgaris	0.2 (0.1)	–	–	–	0.2 (0.1)
Fringilla coelebs^{**L}	105.3 (29.6)	89.0 (29.5)	90.2 (35.6)	81.4 (38.1)	57.3 (21.9)
Serinus serinus	0.4 (0.1)	–	–	–	–
Carduelis spinus	4.4 (1.2)	1.9 (0.6)	3.3 (1.3)	4.1 (1.9)	4.6 (1.8)
C. cannabina	–	–	–	–	0.4 (0.2)
C. flammea	–	0.9 (0.3)	0.4 (0.1)	1.7 (0.8)	3.7 (1.4)
Loxia curvirostra	5.5 (1.5)	2.0 (0.7)	4.6 (1.8)	1.6 (0.7)	3.7 (1.4)
Pyrrhula pyrrhula	4.0 (1.1)	2.9 (0.9)	2.0 (0.8)	1.6 (0.7)	0.9 (0.3)
Coccoth. coccothraustes	0.8 (0.2)	0.1 (0.0)	–	–	0.9 (0.3)
Emberiza citrinella	4.4 (1.2)	0.5 (0.2)	1.5 (0.6)	–	2.2 (0.8)
Total no. of pairs	940	1215	693	751	600

Trends (tested for dominant and influent species only: d>2%): C – curvilinear, L – linear; $*P<0.05$, $**P<0.01$, NS – not significant.

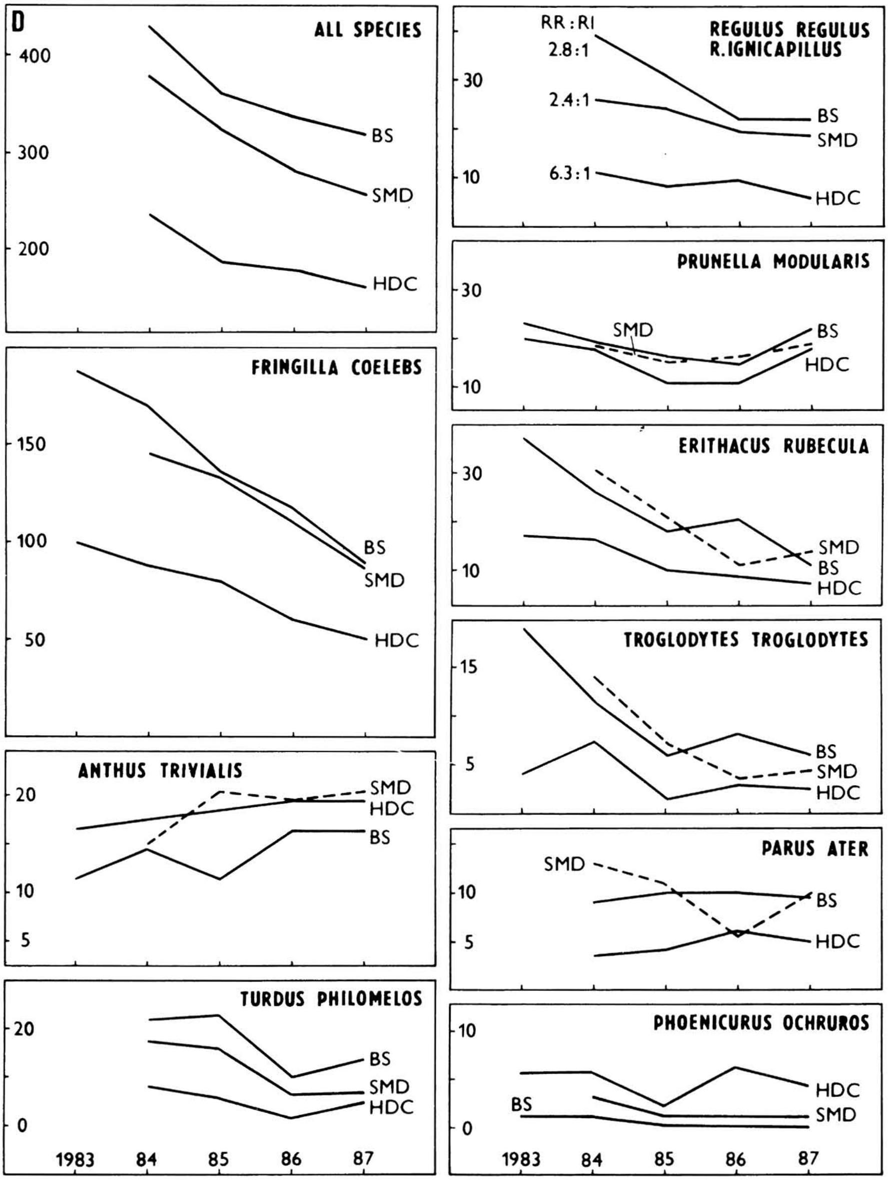

Fig. 1. Line transects: breeding densities (*D*, in pairs/km²) of the most abundant species in different habitats (BS: almost healthy beech-spruce and spruce forests, SMD: slightly and moderately damaged spruce forests, HDC: heavily damaged and dead spruce forests followed by clearcutting) in 1983(84)–1987.

habitats with differing amounts of damage are divided into three groups only. With finer division (the case of point counts), the changes in BSD (the decrease with increasing forest damage) are apparent.

The abundance and structure of bird communities are determined mostly by the vegetation cover, either its structure (MacArthur & MacArthur 1961, Cody 1981, James & Wamer 1982 and others) or its plant species composition (Rotenberry 1985); food availability and also thermoregulation demands (cf. Sabo 1980) can hardly be neglected. Significant changes are apparent in all these factors in the environment affected by industrial emissions (cf. Smith 1981). As forests die out, their structure becomes markedly simpler and plant species composition becomes poorer, local climatic conditions become more extreme and the invertebrate fauna is also secondarily influenced (extreme increase in numbers of one or a few species). The decline of spruce forests in the Krkonoše Mts. leads to their thinning and to the de-

Table 5. Line transects: changes in breeding densities (1984=100%) of the dominant and influent species (d>2%) in 1983–1987 (sample sizes in parentheses, figures rounded to the nearest 0 or 5 for n<200 pairs).

	1983	1984	1985	1986	1987
Anthus trivialis (333)*L	95	100	107	115	117
A. pratensis (46)NS	–	100	65	70	70
Motacilla cinerea (147)NS	95	100	45	55	60
Troglodytes troglodytes (117)*L	110	100	40	45	40
Prunella modularis (300)NS	111	100	71	71	105
Erithacus rubecula (300)**C	118	100	66	54	45
Phoenicurus ochruros (66)NS	100	100	35	85	60
Turdus philomelos (187)*L	170	100	90	30	55
Sylvia atricapilla (228)NS	144	100	132	150	116
Phylloscopus collybita (164)NS	85	100	120	145	70
P. trochilus (133)NS	110	100	85	95	95
Regulus regulus (198)**C	–	100	80	70	60
R. ignicapillus (62)**L	–	100	85	70	55
Parus ater (114)NS	–	100	100	100	100
Fringilla coelebs (1832)**L	112	100	89	71	56
All species (4244)*C	–	100	82	76	69
BS (1305)*C	–	100	84	78	74
SMD (1144)**C	–	100	85	74	68
HDC (1795)*C	–	100	79	76	67

Trends: C – curvilinear, L – linear; *P<0.05, **P<0.01, NS – not significant.

velopment of a very dense and nearly monocultural herb layer. In such conditions declining forest birds (*Regulus, Turdus, Troglodytes, Fringilla, Parus*) are replaced by open habitat species (*Anthus, Saxicola, Phoenicurus*). Decaying forests and newly formed clearings offer conditions for the penetration of some species from the subalpine to the mountain spruce belt, where, in the past, they hade rarely or never bred (*Carduelis flammea*: Flousek 1988, *Anthus pratensis*: Flousek 1987b). Similar observations have been made in the adjacent Jizerské hory Mts. (Kux 1982).

A surprising increase in density with increasing forest damage was discovered in *Motacilla cinerea*, in which a diametrically opposite trend was expected (influence of stream acidity — cf. Ormerod et al. 1985). However, the destruction of spruce forests is correlated with changes in water balance, resulting in soil erosion and the appearence of new water bodies, providing a suitable habitat for this species (Flousek 1987a).

Several authors (Turček 1956, Klíma 1959, Pikula 1967, Bauer & Tichý 1971, Oelke 1980, Christen 1983, Kolbe 1984, Haila et al. 1987; for other references, see Oelke 1980) have dealt with the qualitative and quantitative composition of bird communities of spruce forests in Central Europe. Having compared the densitites obtained in the Krkonoše Mts., the total bird density in good quality stands (BS) varies within the most frequent values quoted by those authors (ca. 300–500 pairs/km² regardless of the methods used, the maxima about 1000 pairs/km²). The spectrum and abundance of dominant species are similar as well (*Fringilla coelebs, Erithacus rubecula, Regulus regulus, Turdus philomelos, Prunella modularis* etc., with the exception of *Parus* spp., whose densities were much lower in the Krkonoše Mts. — cf. Oelke 1980). A comparison with the results from the Polish side of the Krkonoše Mts. leads to similar conclusions (Dyrcz 1973).

Results from the other most heavily affected mountain areas of Czechoslovakia, viz. the Krušné hory Mts. (Šťastný & Bejček 1983, Šťastný et al. 1987) and the Jizerské hory Mts. (Pelc 1986), reveal similar changes in bird communities: an abrupt decrease in breeding bird density, in the number of species and in species diversity with increasing forest damage, and an increase of these in newly formed clearings. There are, however, some variations in the reactions of dominant species to forest damage. While Pelc (1986) found an important correlation between the amount of damage and breeding density for *Fringilla coelebs* and *Regulus regulus* only, Šťastný and Bejček (1983) and this study extend this group to *Regulus ignicapillus, Parus ater, Turdus philomelos* and *Troglodytes troglodytes*. Gramsz (in litt.), on the other hand, reported about the same density of *Fringilla coelebs* in both healthy and almost dead forests on the Polish side of the Krkonoše Mts. The discrepancies, however, may be influenced by the different methods used.

The rapid decrease of breeding bird density in the Krkonoše Mts. during 1983–1987 is remarkable. The decrease could be influenced by a number of factors. However, the trends observed in this study differ markedly from the trends observed over the whole of Czechoslovakia (Šťastný & Janda 1985, Janda et al. in litt.), particularly in *Regulus* spp., *Erithacus rubecula* and *Fringilla coelebs*. On the other hand, the trends are similar in some other species, e.g. *Anthus trivialis, Motacilla cinerea, Phylloscopus* spp. and *Anthus pratensis* (for this species, see also Hudec & Šťastný 1979, Flousek 1987b). The discrepancies may be supposed to be the result of local factors. I suggest that the industrial emissions are the factor most drastically affecting the natural environment of the Krkonoše Mts.

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