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Life span, dispersal and age of nesting Great Grey Owls *Strix nebulosa lapponica* in Sweden

Livslängd, spridning och ålder hos häckande lappugglor Strix nebulosa lapponica *i Sverige*

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Abstract

3073 Great Grey Owls were banded in Sweden in 1955–2012. 416 were controlled at least once (54.6%) or recovered dead (45.4%).Three birds banded as nestlings were recovered in their 17th calendar year. Most birds were recovered during first year of life. Only 4 females were controlled breeding as 2CY birds. 91.3% of birds controlled as first time breeders were at least 4CY. Birds banded as nestlings and recovered dead between September and July moved 100.8 km (mean) with a median distance of 64 km. Juveniles controlled alive moved 45.9 km (mean) with a median distance of 23 km during first year of life. Maximum natal dispersal was 650 km. Median natal dispersal for females was 40 km, between 7 km and 115 km depending on longevity from hatching to control. Females at ages 2CY, 5CY, 8CY and 11CY had the lowest median natal dispersals. Median secondary dispersal distances (from one nest to a later nestsite) for females lay between 0 km and 9 km. For males both mean and median secondary dispersal distances were less than 3 km.

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Introduction

The Great Grey Owl *Strix nebulosa* lapponica has expanded in a south and southwesterly direction since the late 1990s in the northwestern corner of its Palearctic distribution (Ławicki et al. 2013). The expansion southwards in Fennoscandia started between the mid-1960s and early 1990s (Stefansson 1997, Sulkava & Huhtala 1997, Solheim 2009).

Since the first breeding record in Eastern Norway in 1989, the reports of Great Grey Owl sightings gradually increased also in Norway (Solheim 2009). The number of recorded nests or breeding attempts in Norway literally exploded after 2009, resulting in more than 60 nests or breeding attempts recorded in Hedmark county in 2014 (Solheim 2014a, Berg 2015). Since this expansion is closely linked to the Great Grey Owl population in Sweden, we thought a more detailed analysis and presentation of the Swedish banding material of this species would provide valuable background in the attempts to understand this development. The recently published Finnish Bird Ringing Atlas (Valkama et al. 2014) also inspired ideas for some different analysis than formerly presented (Stefansson 1997 and Supplements I-IV, Fransson et al. 2008).

In spite of its considerable size, the Great Grey Owl feeds almost exclusively on small mammals, with *Microtus* voles as the dominant prey species, followed by Bank Vole *Myodes glareolus* and Common Shrew *Sorex araneus* (Mikkola 1983, Stefansson 1997). Their dispersal and movement should thus be expected to follow a pattern similar to what has been found for the microtine specialist Tengmalm's Owl *Aegolius funereus* (Sonerud et al. 1988), and vary according to vole cycles which peak every 3–4 years in the boreal parts of Fennoscandia (e.g. Hanson & Henttonen 1985, Sonerud 1988). The number of recorded breeding Great Grey Owls typically reach a peak at the top of the vole cycles every 3 to 4 years (Mikkola 1983).

Materials and methods

Lists of controlled and recovered Great Grey Owls in Sweden for 1955–1996 were first presented by

Stefansson in the book ”Nordanskogens vagabond” (1997), with four later supplements covering the years 1997–2012. This material supplied with control and recovery data up to and including 2015 make the basis for our new compilations. For comparison with Finland, the number of recoveries up to and including 2012 only (398) has been used. We have not analyzed direction of movement, but concentrated on distance moved and age of owls when controlled or recovered. The formerly presented data has been reviewed and updated to correct for former errors.

15 owls banded in Finland have been controlled alive (9) or recovered dead (6) in Sweden, all within the period 1984–2015. Data from these individuals have been incorporated in the presentation of age of first breeders, and dispersal distances.

Results

From 1955 to 1996 a total of 1715 Great Grey Owls were banded in Sweden (Stefansson 1997). From 1997 to 2012 a total of 1358 Great Grey Owls were banded (Table 1) giving a total of 3073 banded owls as basis for our analysis. For the latter period, the material could be split in birds banded as nestlings (1083; 79.7 %) and adults (275; 20.3 %). A total of 416 individuals were either controlled alive at least once (227; 54.6 %) or recovered dead (189; 45.4 % Figure 1), giving a 13.5% recovery rate. For the period 1997–2012 recoveries have been split to year of banding (Table 1), with recovery rates from each year-class varying from 4.3% to 50.0%. The three years of highest recovery rates (1997, 2000 and 2012) are the years with some of the lowest numbers of banded individuals (13, 9 and 8 individuals respectively). Although only 8 juveniles were banded these years, one of these was actually controlled alive, while none were recovered dead (Table 1). Of 16 birds banded as adults these three years, 3 were later controlled alive while 6 were recovered dead. The overall recovery rate (dead and live controls) from the period 1997–2012 is 10.9%.

Longevity

A total of 122 owls banded as nestlings were later recovered dead, with 3 birds recovered in their 17th calendar year (Figure 2). Most birds were however recovered during their first (33) or second

Table 1. Great Grey Owls banded in Sweden as nestlings or adults in 1997–2012 (B), and later controlled alive (C) or recovered dead (D).
Lappugglor ringmärkta som boungar eller vuxna häckfåglar i Sverige 1997–2012 (B), och senare kontrollerade (C) eller återfunna döda (D).

Year	Banded as nestlings *Märkta som ungar*			Banded as adults *Märkta som vuxna*			Total			% of total *% av total*		
	B	C	D	B	C	D	B	C	C	C	D	C+D
1997	0	0	0	13	0	3	13	0	3	-	23.1	23.1
1998	73	6	5	24	2	4	97	8	9	8.2	9.3	17.5
1999	44	5	2	9	1	1	53	6	3	11.3	5.7	17.0
2000	6	1	0	3	2	0	9	3	0	33.3	-	33.3
2001	89	7	0	21	8	3	110	15	3	13.6	2.7	16.4
2002	69	1	1	15	3	1	84	4	2	4.8	2.4	7.1
2003	21	0	1	3	0	1	24	0	2	-	8.3	8.3
2004	193	7	11	40	11	3	233	18	14	7.7	6.0	13.7
2005	78	0	3	12	0	1	90	0	4	-	4.4	4.4
2006	17	1	0	18	2	2	35	3	2	8.6	5.7	14.3
2007	111	3	4	32	4	1	143	7	5	4.9	3.5	8.4
2008	53	0	2	15	1	1	68	1	3	1.5	4.4	5.9
2009	1	0	0	7	1	0	8	1	0	12.5	-	12.5
2010	121	2	2	29	10	4	150	12	6	8.0	4.0	12.0
2011	205	2	6	28	2	0	233	4	6	1.7	2.6	4.3
2012	2	0	0	6	1	3	8	1	3	12.5	37.5	50
Sum	1083	35	37	275	48	28	1358	83	65	6.1	4.8	10.9

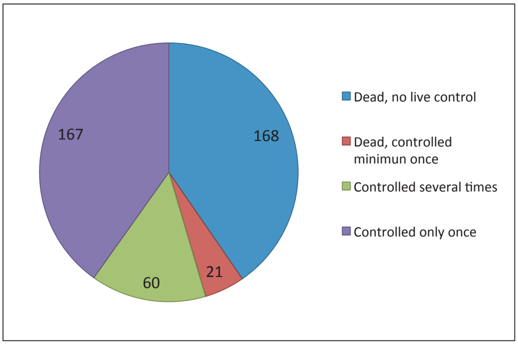

Figure 1. Recoveries and controls of 416 Great Grey Owls banded in Sweden.
Återfynd och kontroller av 416 lappugglor ringmärkta i Sverige.

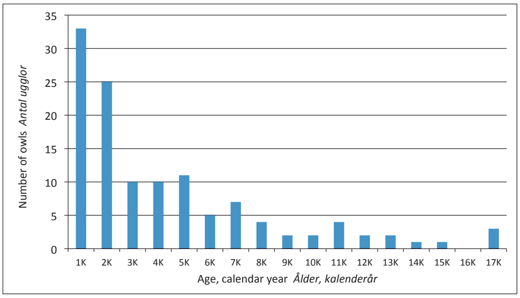

Figure 2. Age at recovery of dead Great Grey Owls banded as nestlings.
Ålder vid återfynd av döda lappugglor ringmärkta som boungar.

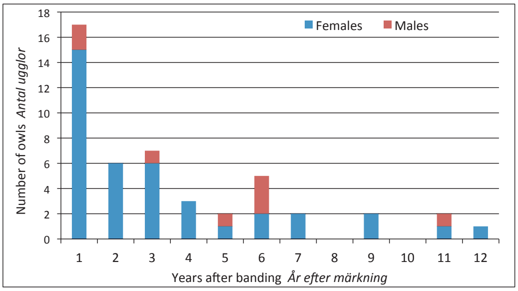

Figure 3. Years after banding for recoveries of dead Great Grey Owls banded as nesting adults.
År efter märkning för återfynd av döda lappugglor ringmärkta som vuxna häckfåglar.

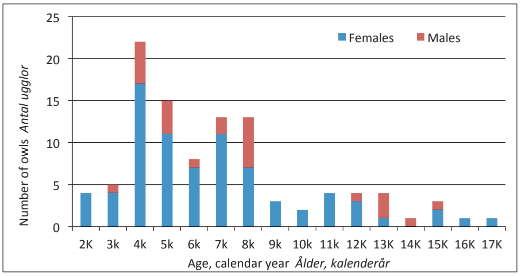

Figure 4. Age of Great Grey Owls banded as nestlings when first controlled as nesting adults.
Ålder vid förstagångskontroll som häckande fågel för lappugglor märkta som boungar.

(25) calendar years, with numbers recovered steadily declining thereafter. Recovery of dead birds banded as nesting adults comprise of 39 females and 8 males, which also show a declining number of recoveries with age (Figure 3). The oldest bird from this group was a female recovered 12 years after banding. Her exact age is however not known, since aging of fully grown Great Grey Owls according to molt patterns (Suopajärvi & Suopajärvi 1994, Solheim 2011) was not well known at the time for banding of most of the owls in this material.

Age of first time breeders

Seventy-six females and 27 males banded as nestlings were later controlled as nesting birds at least once (Figure 4). Most birds were controlled for the first time in their fourth calendar year or later. Four females were found breeding as one year old birds (2K), and 4 females and 1 male as two year olds (3K). The oldest first time control breeder banded as a nestling was a female controlled in her 17th calendar year (16 years old). Another female banded as nestling was controlled breeding as 5K, 10K, 16K and finally as 17K bird. Of all controlled nesting birds themselves banded as nestlings, 91.3% were at least three years old (4K+) when controlled nesting for the first time. The figure shows a tendency for more birds controlled at age classes 4K-5K, 7K-8K and 11K–12k (13K).

Dispersal

Juveniles

Fifty of a total of 52 juveniles recovered dead or controlled alive between 1 August and 30 June the following year dispersed less than 250 km from where they were banded as nestlings (Figure 5). One individual was found dead 410 km from the hatch site, and one was shot 420 km away in Russia. Birds recovered before the end of September were mostly found very close to their hatch site. Mean distance for 29 birds found dead after Sep-

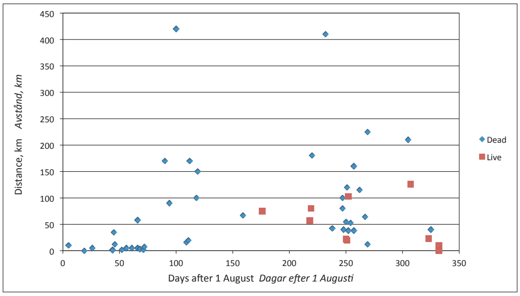

Figure 5. Distance in km and days after 1 August at time of recovery or control of Great Grey Owls banded as nestlings.
Avstånd i km och dagar efter den 1 augusti vid tiden för återfynd eller kontroll av lappugglor ringmärkta som boungar.

tember, was 100.8 km, with a median distance of 64 km.

Ten juveniles were controlled alive between 23 January and 2 June in their second calendar year. Two individuals were controlled 126 km and 103 km from the hatch site respectively, while the rest were all controlled within 80 km away. Mean distance moved for these controlled birds was 45.9 km, with a median distance of 23 km.

Full-grown birds

A total of 103 birds banded as nestlings were later controlled as breeders (76 females and 27 males). The maximum natal dispersal recorded was a female banded in 1982 and controlled in 1987 (6K), which moved 650 km. Next in line was another female controlled as 13K after dispersing 230 km. Mean distance of natal dispersal for all adults is shown linked to age of the bird at first control (Figure 6), and varied between 29.9 km and 114.4 km.

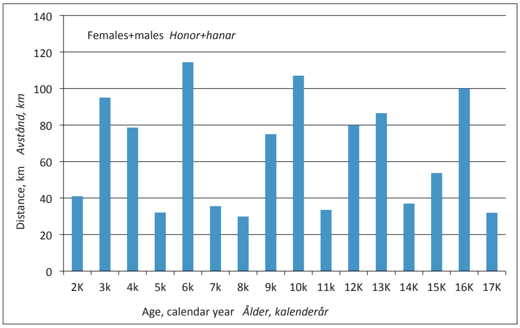

Figure 6. Mean distance in km between hatch site and nest site when controlled as nesting for the first time, female and male pooled.
Medelavstånd i km mellan kläckningsplats och platsen för första kontroll som häckfågel för lappugglor av båda könen.

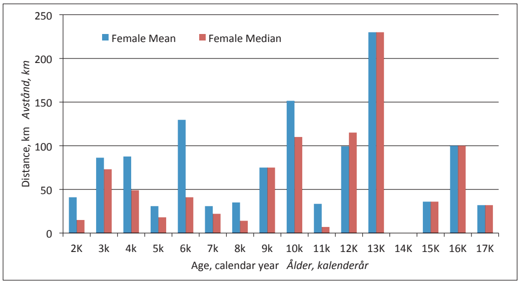

Figure 7. Mean and median distance in km between hatch site and nest site for Great Grey Owl females when first controlled as nesting birds.
Medel- och medianavstånd i km mellan kläckningsplats och boplats vid förstagångskontroll som häckfågel för lappugglehonor.

For females only, the mean natal dispersal varied between 30.8 km (7K) and 151.5 km (10K), with an overall mean of 72.3 km (Figure 7). Median values for the females' natal dispersal fell between 7 km and 115 km, with an overall median of 40 km. The lowest median natal dispersal values for Swedish Great Grey Owl females are for birds recorded nesting for the first time in age classes 2K, 5K, 8K and 11K respectively.

A total of 135 females (26 banded as juveniles and 109 banded as breeders) were controlled as breeders more than once, some up to 6 times, resulting in a total of 211 controls giving distances between first recorded nest place and later nest places (Figure 8), and covering 1 to 12 years af-

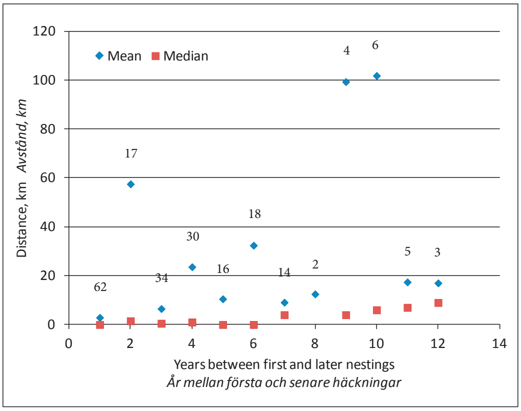

Figure 8. Mean and median distance dispersed in km between first recorded nesting of females and later nest sites. Number of controlled females is given at each symbol.
Medel- och medianavstånd i km mellan den första registrerade häckningsplatsen för lappugglehonor och deras senare häckningsplatser. Antal kontrollerade honor anges vid varje symbol.

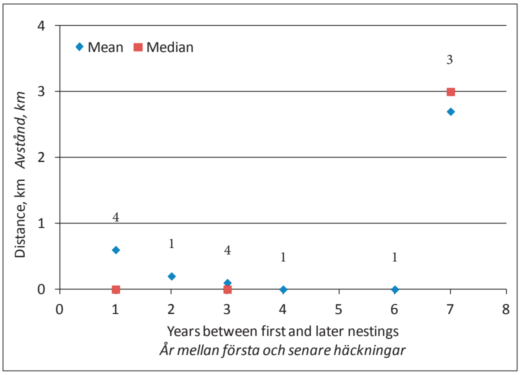

Figure 9. Mean and medain distance dispersed in km between first recorded nesting of males and later nestsites. Number of controlled males is given at each symbol.
Medel och median avstånd i km mellan den första registrerade häckningsplatsen för lappugglehanar och deras senare häckningsplatser. Antal kontrollerade hanar anges vid varje symbol.

ter first breeding record. Mean dispersal distance moved after first having nested once varies between 2.9 km and 101.8 km, with an overall mean dispersal distance of 19.6 km. Median secondary dispersal distances however are very low, varying between 0 and 9 km. Only 9 males were controlled as breeders after their first control as nesting birds, resulting in a total of 14 controls only. Both mean and median secondary dispersal distances for males are below 3 km (Figure 9), spanning 1 to 7 years after the first breeding record.

Discussion

Recovery and control rates

The number of Great Grey Owls banded in Sweden 1911–2012 (3091; Fransson et al. 2013) is next to identical to the numbers banded in Finland 1913–2012 (3110; Valkama et al. 2014). The Swedish material however contains almost twice as many recovered and controlled individuals (398 versus 201). The number of birds controlled alive in Sweden is 3.8 times as high as in the Finnish material (262 versus 69). For birds banded as nestlings and later controlled as breeders the difference is even higher, with 97 individuals controlled in Sweden and only 18 controlled in Finland. These discrepancies can only be explained by a higher effort at catching and controlling fully grown birds in Sweden. The higher number of birds recovered dead of Great Grey Owls banded in Sweden (189 versus 123) may be linked to Sweden having almost twice as many inhabitants as Finland (9.8 versus 5.5 million), thus providing higher probabilities for a dead owl to be found.

Recoveries versus vole cycles

The highest recovery rates were found for birds banded in typical vole-low years when only a very low number of birds were banded in total (8, 9 and 13). One might expect a higher probability of finding starved individuals in such years, so it is surprising that the only bird recovered and banded as a nestling from these years was actually controlled as a nesting adult two years after being banded. The adults however follow the expected pattern, with 6 individuals recovered dead and 3 controlled alive out of 16 banded birds.

The distribution of birds banded as nestlings and later recovered dead followed an expected pattern, with most individuals recovered during their first autumn after hatching, and steadily declining thereafter (Figure 2). Since most nestlings are banded in vole peak years (but some also in the previous pre-peak years), they will usually encounter a vole depression as 2K (or 3K) birds. Many will meet a slightly increasing vole abundance as 3K and definitively as 4K birds. The low numbers of dead birds in age classes 3K and 4K may thus be explained by a higher survival rate for birds which have survived their first two years of life. Even birds banded as nesting adults and later recovered dead are most likely found during the first year after being banded (Figure 3). The total number of recovered adults is however too small to draw any other conclusion than that the oldest one lived 12 years after banding. Since most birds banded as nestlings and later controlled as nesting birds were at least in age group 4K or older (Figure 4), this individual may well have lived to be 16 years. This is in accordance with the age of the oldest nesting bird controlled for the first time (Figure 4), and matches the age record in the Finnish material (16 years; Valkama et al. 2014).

Age of nesting birds

The controls of nesting birds banded as nestlings show that at least some females may start breeding as one year old birds (Figure 4), and at least one male started as two years old. Cramp (1985) gives the same ages of first breeders, quoting Stefansson with age of first breeding probably 2 years, but one female recorded breeding as 1 year old bird (2K).

The highest number of birds controlled for the first time as nesting adults fell in the age category 4K. This is in accordance with a vole cycle peaking after 3 years. Interestingly the age distribution peaks again for birds in age catergories 7K–8K, and even tend to show a peak for ages 11K–13K. The numbers for these latter age groups are however small, and may be influenced by accidential events. The peaks for age groups 4K and 7K–8K however follows what would be expected from vole cycles peaking every 3 to 4 years.

Natal dispersal

In Finland most juveniles recovered during their first year were found less than 50 km from their hatch site. Juveniles later controlled alive were however encountered closer to their hatch site than the ones recovered dead (Figure 5). This might be caused by these individuals being better hunters and survivors close to their hatch sites when voles decline, but could also be linked to bird banders being more active at catching and controlling birds locally where they band them as nestlings.

Birds banded as nestlings and later controlled as nesting adults showed mean values of natal dispersal between 30 km and 144 km (Figure 6), which is in the same magnitude as for juveniles recovered dead during first year of life. Median natal dispersal for Finnish birds banded as nestlings was 48 km for females (N=13) and 25 km for males (N=5) (Valkama et al. 2014). This is in accordance with an overall median natal dispersal distance for Swedish female owls of 40 km. The lowest median natal dispersal distances found for females in age classes 2K, 5K, 8K and 11K (Figure 7) can also be interpreted as a result of vole cycles peaking every 3rd year, as found for northern Sweden (Hipkiss et. al 2008). The females breeding as 2K birds probably encounter a vole peak the year after they were hatched, and thus disperse short distances from their hatch site. The shorter distances for later age classes may be a result of birds returning to their hatch site or former breeding site during vole peak years.

Breeding dispersal

When the owls have nested for the first time, both females (Figure 8) and males (Figure 9) are almost stationary regarding nest site, and show median dispersal values to next breeding site mostly close to zero, with maximum 9 km for females and 3 km for males. 34 females in Finland which were controlled in two successive breeding seasons also moved less than 10 km (Valkama et al. 2014). While Swedish females seem to breed in the same area even up to 12 years after the first control (maximum distance moved was 102 km), Finnish females showed median movement distances of 67 km with a maximum of 405 km moved.

Site tenasity or nomadism

During the microtine low in 2012 which covered most of Fennoscandia, at least 76% of 144 individuals identified from photographs were juvenile 2K birds (Solheim 2014b). Older birds do however also appear along the coast of southern Norway and Sweden (Solheim 2010), leaving the question open to whether adult birds stay close to their breeding sites during winter, or migrate to better hunting grounds before returning to the breeding areas they have used at least once. To get more data on winter movements, a satellite project was initiated in 2014 and two nesting females were equipped with transmitters (Solheim et al. 2015). While one of them stayed within 26 km of the nest place of 2014 from February until loosing its transmitter in late May 2015, the other moved to a new locality 175 km away where it stayed for one year until the transmitter went silent in February 2016. Hopefully more birds with transmitters will give better data on the Great Grey Owls winter habits in the future.

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Sammanfattning

Lappugglan *Strix nebulosa lapponica* har expanderat i en sydlig och sydvästlig riktning sedan sent 1990-tal i det nordvästra hörnet av sin palearktiska utbredning (Ławicki et al. 2013). Expansionen söderut i Fennoskandia började mellan mitten av 1960-talet och början av 1990-talet (Stefansson 1997, Sulkava & Huhtala 1997, Solheim 2009). Efter den första kända häckningen i östra Norge 1989 ökade rapporterna om observationer av lappugglor även i Norge (Solheim 2009). Antalet registrerade bon med häckningar eller häckningsförsök formligen exploderade i Norge efter 2009 vilket resulterade i mer än 60 bon med häckande lappugglor i Hedmark fylke 2014 (Solheim 2014a, Berg 2015). Eftersom denna expansion är nära sammankopplad med lappugglans uppträdande och förekomst i Sverige tänkte vi att en mer detaljerad analys och presentation av det svenska ringmärkningsmaterialet skulle kunna ge värdefull bakgrundsinformation i försöken att förstå denna utveckling. Den nyligen publicerade Finnish Bird Ringing Atlas (Valkama et al. 2014) inspirerade till andra analyser än tidigare presenterade (Stefansson 1997 och supplementen 1–4, 2001–2013, Fransson et al. 2008).

Trots sin imponerande storlek lever lappugglan nästan uteslutande på små däggdjur med sorkar *Microtus* som det dominerande bytesdjuret följt av långsvansad skogssork *Myodes glareolus* och vanlig näbbmus *Sorex araneus* (Mikkola 1983, Stefansson 1997). Deras spridning och rörelser bör således förväntas följa ett mönster som liknar vad som hittats hos sorkspecialisten pärlugglan *Aegolius funereus* (Sonerud et al. 1988) och som varierar beroende på sorkcyklerna med toppar vart 3–4 år i de boreala delarna av Fennoskandia (e.g. Hanson & Henttonen 1985, Sonerud 1988). Antalet registrerade häckningar av lappugglor blir följdriktigt störst vid toppen av sorkcyklerna vart 3–4 år (Mikkola 1983).

Material och metoder

Förteckningar över kontrollerade (levande) och återfunna (döda) lappugglor i Sverige presenterades först av Stefansson i boken ”Nordanskogens vagabond” (1997) och senare med fyra supplement för åren 1997–2012. Detta inklusive material med kontroll- och återfyndsdata till och med år 2015 utgör grunden för våra sammanställningar. Vi har inte analyserat spridningsriktning utan koncentrerat oss på flyttningsavstånd och fåglarnas ålder vid kontrollerna och återfynden. Tidigare presenterade data har granskats och uppdaterats för att korrigera förekommande fel. Femton lappugglor ringmärkta i Finland har kontrollerats (9) eller återfunnits döda (6) i Sverige samtliga under 1984–2015. Data från dessa individer har införlivats i presentationen av ålder vid första häckning och spridningsavstånd.

Resultat

Från 1955 till 1996 har totalt 1715 lappugglor ringmärkts i Sverige (Stefansson 1997). 1997–2012 blev sammanlagt 1358 ringmärkta (Tabell 1) vilket ger totalt 3073 ringmärkta lappugglor som grund för vår analys. Av dessa har totalt 416 individer kontrollerats vid ett eller flera tillfällen (227, 54,6%) eller återfunnits döda (189; 45,4% Figur

1) vilket ger en återfyndsgrad av 13,5%. För den senare perioden har materialet delats upp i fåglar ringmärkta som boungar (1083, 79,7%) och vuxna (275, 20,3%). För perioden 1997–2012 har återfynden delats upp i år de ringmärktes (Tabell 1) med återfyndsprocent för varje årsklass varierande från 4,3% till 50,0%. De tre åren med högst återfyndsprocent (1997, 2000 och 2012) är år med några av det lägsta antalet ringmärkta individer (13, 9 respektive 8). Även om endast åtta ungar ringmärktes dessa år kontrollerades faktiskt en av dem levande medan ingen återfanns död (Tabell 1). Av 16 ringmärkta som vuxna under dessa tre år kontrollerades tre senare levande medan sex återfanns döda. Den totala återfyndsgraden (döda och levande kontroller) från perioden 1997–2012 är 10,9%.

122 lappugglor ringmärkta som ungar återfanns senare döda varav tre under sitt 17:e kalenderår (Figur 2). De flesta återfanns emellertid under deras första (33) eller andra (25) kalenderår medan antalet återfunna stadigt minskande därefter. Återfynden av döda lappugglor ringmärkta som häckande (vuxna) bestod av 39 honor och 8 hanar som också visar minskande antal återfynd med åldern (Figur 3). Den äldsta fågeln från denna grupp var en hona som återfanns 12 år efter ringmärkningen. Den exakta åldern är emellertid inte känd eftersom åldern hos vuxna lappugglor enligt ruggningsmönster (Suopajärvi & Suopajärvi 1994, Solheim 2011) inte var känd vid tidpunkten för ringmärkningen vilket gäller de flesta vuxenmärkta ugglorna i detta material.

76 honor och 27 hanar ringmärkta som ungar kontrollerades senare som häckande minst en gång (Figur 4). De flesta kontrollerades för första gången i deras fjärde kalenderår eller senare. Fyra honor påträffades häckande som ettåriga (2K) och fyra honor samt en hane som tvååriga (3K). Den äldsta som påträffades häckande för första gången var en hona som kontrollerades i sitt 17:e kalenderår (16 år gammal). 91,3% av alla kontrollerade var häckande fåglar ringmärkta som ungar och minst tre år gamla (4K+) när de kontrollerades häckande för första gången. Figur 4 visar en tendens till flera toppar av fåglar som kontrollerats i åldersklasserna 4K–5K, 7K–8K och 11K–12K (13K).

Ungfåglarnas spridning redovisas i Figur 5. 50 av totalt 52 unga lappugglor återfanns döda eller kontrollerades mellan 1 augusti och 30 juni följande år mindre än 250 km från där de ringmärktes som ungar. En av dem påträffades död 410 km från födelseplatsen och en sköts i Ryssland 420 km bort. Fåglar återfunna före utgången av september hittades mestadels mycket nära födelseplatsen. Det genomsnittliga avståndet för 29 fåglar som påträffades döda efter september var 100,8 km med ett medianavstånd av 64 km. 10 ungfåglar kontrollerades mellan den 23 januari och 2 juni i deras andra kalenderår. Två av dem kontrollerades 126 km respektive 103 km från födelseplatsen medan övriga kontrollerades inom 80 km. Det genomsnittliga flyttningsavståndet för dessa kontrollerade fåglar var 45.9 km, med ett medianavstånd av 23 km.

De adulta fåglarnas spridning visas i Figur 7–9. 103 lappugglor ringmärkta som ungar kontrollerades senare som häckande (76 honor och 27 hanar). Den maximala natalspridningen, alltså för fåglar registrerade som häckande för första gången, som registrerats var en hona ringmärkt 1982 och kontrollerad 1987 (6K) och som flyttat 650 km. Nästa i raden var en annan hona kontrollerad som 13K efter 230 km. Det genomsnittliga avståndet natalspridning för alla vuxna visas länkade till fågelns ålder vid första kontrollen (Figur 6) och varierade mellan 29,9 km och 114,4 km. För enbart honor varierar natalspridningen mellan 30,8 km (7K) och 151,5 km (10K) med ett medelvärde av 72,3 km (Figur 7). Medianvärdet för honornas natalspridning låg mellan 7 km och 115 km med ett medelvärde av 40 km. Det lägsta värdena för natalspridning hos de svenska lappugglehonorna fanns i åldersklasserna 2K, 5K, 8K och 11K.

Totalt 135 honor (26 ringmärkta som ungar och 109 som häckande) kontrollerades som häckande mer än en gång, vissa upp till 6 gånger, vilket resulterar i totalt 211 kontroller av avstånd mellan deras första och senare boplatser (Figur 8) och omfattar 1–12 år efter den första registrerade häckningen. Medelvärdet av spridningsavståndet efter att ha häckat första gången varierar mellan 2,9 km och 101,8 km med en genomsnittlig spridning av 19,6 km. Medianvärdet för den sekundära spridningen är mycket lågt och varierande mellan 0 och 9 km. Endast nio hanar kontrollerades som häckande efter den första kontrollen som häckande vilket resulterade i endast 14 kontroller. Både medelvärdet och medianvärdet av sekundärt spridningsavstånd för hanar är mindre än 3 km (Figur 9) inom spannet 1–7 år efter den första registrerade häckningen.

Diskussion

Antalet lappugglor ringmärkta i Sverige 1911–2012 (3091; Fransson et al. 2013) är närmast identisk med antalet ringmärkta i Finland 1913–2012 (3110; Valkama et al. 2014). Det svenska materialet innehåller emellertid nästan dubbelt så många återfunna och kontrollerade individer (398 respektive 201). Antalet fåglar kontrollerade i Sverige t.o.m. 2012 är nästan fyra gånger så stort som i det finska materialet (262 kontra 69). För fåglar ringmärkta som ungar och senare kontrollerade som häckande är skillnaden ännu större med 97 individer som kontrollerats i Sverige och endast 18 i Finland. Dessa skillnader kan endast förklaras av en högre ansträngning på att fånga och kontrollera vuxna lappugglor i Sverige. Det högre antalet återfunna döda utav svenskmärkta lappugglor (189 kontra 123) kan kopplas till att Sverige har nästan dubbelt så många invånare än Finland (9,8 kontra 5,5 miljoner), vilket ger högre sannolikheter för att en död lappuggla påträffas.

De högsta återfyndsprocenten finns bland lappugglor ringmärkta under typiska bottenår hos sorkarna, år när endast ett litet antal fåglar ringmärktes (8, 9 och 13). Man kan förvänta sig en högre sannolikhet för att svältande individer påträffas under sådana år. Det är därför förvånande att den enda återfunna fågeln från dessa år var en ringmärkt som unge och som faktiskt kontrollerades häckande två år senare. De vuxna följer däremot förväntade mönster med sex återfunna döda och tre kontrollerade individer av 16 ringmärkta.

Spridningen av lappugglor ringmärkta som ungar och senare återfunna döda följde ett förväntat mönster med de flesta individer återfunna under sin första höst efter kläckningen och antalet stadigt minskande därefter (Figur 2). Eftersom de flesta ungar är ringmärkta under sorkarnas toppår (men några även i det föregående uppgångsåret), möter de som regel en sorkkrasch som 2K (eller 3K) fåglar. Många möter en begynnande sorktillväxt som 3K fåglar och märkbart som 4K fåglar. Det låga antalet döda fåglar i åldersklasserna 3K och 4K kan således förklaras av en högre överlevnad hos fåglar som har överlevt de första två åren av sitt liv. Även fåglar ringmärkta som häckande vuxna och som senare återfunnits döda hittas mest sannolikt under det första året efter ringmärkningen (Figur 3). Det totala antalet återfunna vuxna är dock för litet för att kunna dra någon annan slutsats än att den äldsta överlevt 12 år efter ringmärkningen. Eftersom de flesta fåglar ringmärkta som ungar och senare kontrollerats som häckande fåglar var minst i åldersklassen 4K eller äldre (Figur 4) kan denna individ mycket väl ha levt till 16 år. Detta är i enlighet med åldern hos den äldsta häckande lappugglan som kontrollerades för första gången (Figur 4) och matchar åldersrekordet i det finska materialet (16 år; Valkama et al. 2014).

Kontrollerna av häckande lappugglor ringmärkta som ungar visar att åtminstone några honor häckade redan som ettåriga (Figur 4) och minst en hane vid två års ålder. Cramp (1985) anger att den genomsnittliga åldern för första häckningen inte är känd men refererar till Stefansson om ålder hos den första registrerade häckningen för en hona häckande som ett år gammal fågel (2K).

Det högsta antalet som kontrollerats för första gången som vuxna finns i ålderskategorin 4K. Detta överensstämmer med en sorkcykeltopp efter tre år. Intressant är också topparna som återkommer för fåglar i ålderskategorin 7K–8K och som även tenderar att visa en topp för åldrarna 11K–13K. Siffrorna för dessa sistnämnda åldersgrupper är dock små och kan påverkas av tillfälliga händelser. Topparna för åldersgrupperna 4K och 7K–8K följer emellertid vad som kan förväntas vid sorkcykler med toppår vart 3–4 år.

I Finland återfanns flest ungfåglar under deras första år mindre än 50 km från deras födelseplats. Ungfåglar som senare kontrollerades påträffades emellertid närmare deras födelseplats än de som återfanns döda (Figur 5). Detta kan ha orsakats av att dessa individer var bättre utvecklade som jägare eller att tillgången på bytesdjur var förhållandevis god nära födelseplatsen när sorktillgången senare gick ner men kan också kopplas till att ringmärkaren var mer aktiv med att fånga och kontrollera fåglar i närheten av lokalerna där de ringmärkte dem som ungar.

Fåglar ringmärkta som ungar och senare kontrollerade som häckande visar medelvärden av natalspridning mellan 30 km och 144 km (Figur 6) vilket är i samma storleksordning som för ungfåglar återfunna döda under första levnadsåret. Medianvärdet för natalspridning för finska fåglar ringmärkta som ungar var 48 km för honor (n = 13) och 25 km för hanar (n = 5). Detta är i enlighet med ett övergripande median natalspridningsavstånd på 40 km för svenska honor. Det lägsta median natalspridningsavståndet hos honor i åldersklasserna 2K, 5K, 8K och 11K (Figur 7) kan också tolkas som ett resultat av sorkcyklernas topp vart tredje år, som dokumenterats för norra Sverige (Hipkiss et. al 2008). Honorna som häckar som 2K fåglar möter då förmodligen en sorktopp året efter att de har kläckts och sprider sig därför kortare avstånd från födelseplatsen. Därefter kan det vara så att honorna vänder tillbaka till trakterna där de kläcktes eller tidigare har häckat.

När ugglorna har häckat för första gången är både honor (Figur 8) och hanar (Figur 9) ofta trogna häckningsplatsen eller har avstånd till nästa häckningsplats på maximalt 9 km för honor och 3 km för hanar. 34 honor i Finland som kontrollerades i två på varandra följande häckningssäsonger flyttade också mindre än 10 km. Medan svenska honor påträffats häcka i samma område upp till 12 år efter den första kontrollen (maximalt flyttavstånd var 102 km) visade finska honor en median spridning av 67 km med högst 405 km förflyttning.

Under den mycket dåliga sorktillgången 2012 som täckte större delen av Fennoskandia var minst 76% av 144 individer 2K-fåglar identifierade från fotografier av lappugglor i Sverige (Solheim 2014b). Äldre fåglar visar sig emellertid också regelmässigt längs kusten i södra Norge och Sverige (Solheim 2010) vilket lämnar frågan öppen om huruvida gamla fåglar stannar i närheten av sina häckningsplatser under vintern eller migrerar mer eller mindre långt till andra eventuellt bättre jaktmarker innan de evantuellt återvänder till häckningsområden de har använt minst en gång. För att få mer data på vinterrörelser inleddes ett satellitprojekt år 2014 och två i östra Norge häckande honor utrustades med sändare (Solheim et al. 2015). En av dem uppehöll sig inom 26 km från 2014 års häckningsplats från februari 2015 tills den förlorade sändaren i slutet av maj samma år. Den andra flyttade till en ny lokal 175 km bort där den stannade i ett år tills sändaren tystnade i februari 2016. Förhoppningsvis ger fler fåglar med sändare bättre data om lappugglornas vintervanor i framtiden.

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Does the breeding performance differ between solitary and colonial breeding Red-throated Loons *Gavia stellata* in the core area of the Swedish population?

Skiljer sig häckningsutfallet mellan ensam- och kolonihäckande smålommar Gavia stellata *i artens svenska kärnområde?*

BÖRJE DAHLÉN & MATS O. G. ERIKSSON

Abstract

Most Red-throated Loons *Gavia stellata* are solitary breeders in small pools and provide fish prey to the chicks from larger waters at a distance seldom exceeding 10 km. More rarely, several pairs nest together in colonies. We compared the breeding performance of solitary and colonial breeders in South-central Sweden during 2000–2016 (except for 2007). Annual productivity was 0.63 and 0.52 chicks per pair and year for solitary and colonial breeding pairs respectively, and mean percentage per year of broods with two chicks did not differ, 24% and 18% (no significant differences). Average hatching probability was the same, 0.576 and 0.581 over years. The average probability however, that at least one chick would be fledged was significantly higher among solitary pairs, 0.915 and 0.731 respectively, although the overall probability of successful breeding did not differ, 0.530 and 0.443. Thus, any benefit with reference to coloniality can be questioned. More likely, colonial breeding was a result of habitat selection, with a concentration of several nesting pairs within a relatively small area with a profitable foraging lake nearby.

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Colonial breeding among birds, defined as an assemblage of nests within a limited area and with the primary source of food outside the territory, has been intensively studied in several bird species (e.g. Brown & Brown 2001 for an overview). Nevertheless, it remains an elusive phenomenon which is hard to explain. With the development of the behavioural ecology approach to explain the evolution of various adaptations in terms of a balance between fitness-related benefits and costs during the 1970s, studies of coloniality were primarily focused on predation risk and finding food (e.g. Chapter 2 in Perrins & Birkhead 1983 for an overview). Avoiding predation was thought to occur either actively by increased vigilance (several pairs of eyes see more than a few) and mobbing of predators, or passively by dilution (although a colony may attract predators, as long as the predation pressure does not increase in proportion to the number of nests in the colony, the risk for any individual to fall victim will decrease). With reference to obtaining food, colonies have been proposed to serve as ‘information centres’, where individual birds may benefit from observing and following conspecifics to assumed food resources. Over time however, the theoretical approach shifted from individual benefits to interpreting coloniality as the result of, or a by-product of, so called commodity selection with reference to the availability of e.g. habitats or mates (e.g. Danchin & Wagner 1997, Wagner et al. 2000). Based on a comparative study including 320 bird species, colonial breeding has been proposed to be correlated with the absence of a feeding territory, breeding in aquatic habitats, and exposure to predators (Rolland et al. 1998).

The Red-throated Loon *Gavia stellata* fits into this categorisation in so far as it is linked to aquatic habitats, forages outside the breeding territory, and that nest predation is a key reason for breeding failure (e.g. Dickson 1993, Eberl & Picman 1993, Dahlén & Eriksson 2002, Rizzolo et al. 2014). However, it most often breeds solitarily with single pairs nesting in small pools, often smaller than 1 ha and rarely larger than 10 ha, and mostly without fish. Prey fish for the non-fledged chicks are caught from larger lakes with clear water, or at sea, with the travelling distance rarely exceeding 10 km (e.g. Eriksson 2010). Less often, several nesting Red-throated Loon pairs may group together in colonies (e.g. Chapter 11 in Lack 1968, Cramp & Simmons 1977).

In this study, we compare the breeding success of solitary breeding pairs with pairs breeding in a colony in the same geographical region in South-central Sweden, in order to investigate if there is any indication of different breeding performance among solitary and colonial breeding pairs. No previous comparisons of this character, as far as we know, have been published with reference to the Red-throated Loon.

Study area

The study area is located in the County of Dalarna within the core area of the Swedish population of Red-throated Loon. Dalarna together with the neighbouring counties (Värmland, Örebro and Västmanland) harbour a total of 470–600 pairs, or about one third of the total population in Sweden (data from Ottosson et al. 2012). Prey fish for the chicks are almost exclusively caught in nutrient-poor freshwater lakes with a dominance of small-sized cyprinid or salmonid fish (Eriksson 2006, Eriksson & Paltto 2010).

The breeding performance of solitary breeding pairs was investigated in the same area as in a previous study of the target species in Malung-Sälen Municipality (60º41’N, 13º43’E) in the County of Dalarna (Figure 1 in Dahlén & Eriksson 2002). The area is approximately 1,820 km^2, with a population of around 70 Red-throated Loon pairs that have been surveyed on an annual basis during the period 1991–2016. During 1991–2002, the surveys covered 130–150 pools or the majority of potential breeding sites, but after that the annual field surveys have been restricted to a smaller sample of 27–91 pools. The presence of a territory-holding pair has been recorded in at least one year in 99

Figure 1. Breeding pool regularly used by a solitary Red-throated Loon *Gavia stellata* pair in the study area. Photo: Börje Dahlén.
Häckningstjärn som regelbundet nyttas av ett ensamhäckande smålomspar. Foto: Börje Dahlén.

Figure 2. The Tisjökölen mire area, used by colonial breeding Red-throated Loons *Gavia stellata*, as viewed from the observation tower. Photo: Börje Dahlén.
Myrområdet Tisjökölen, vy från fågeltornet. Foto: Börje Dahlén.

pools (Figure 1 shows one of the sites). 15–20 nutrient-poor clear-water lakes are used for foraging, with distances ranging from less than 0.5 km to 4.6 km between breeding pools and foraging lakes. For the purpose of this study, results from the field surveys during the period 2000-2016 have been used, with the exception of 2007 when survey coverage was too poor.

In parallel, the breeding performance of a concentration of Red-throated Loons breeding at Tisjökölen (60º54'N, 13º6'E; Figure 2) was studied. This is a mire site of ca 0.85 km^2 and is located approximately 35 km northwest of the centre of the above described area (pages 120–122 in DOF 2005 for a further description). The Red-throated Loon population has been surveyed in the period 2000–2016 (except for 2007), and the number of pairs has fluctuated at around 20, making the colony at Tisjökölen the largest known concentration of breeding Red-throated Loon at a single site in Sweden. The distance between single nests can be as little as 30–40 meters, and 2–3 pairs may breed within the same pool. The birds catch fish prey for the chicks at Lake Tisjön, located immediately north of the mire area and at a distance of ca 1 km from the centre of the mire to the nearest shore of the lake.

Previous studies (Dahlén & Eriksson 2002) indicate that the nest initiation in the study area is spread over a period of approximately six weeks, from mid-May until the end of June, although around half of the nests are initiated over a concentrated period of around ten days during early or mid-June. Around 15% of failed clutches are followed by replacement clutches initiated from early June until mid-July. Very few chicks are thus hatched after the mid or end of August, when there is still a period of 2-3 months before ice-cover; i.e. freezing does not constitute a breeding constraint.

Methods

Field surveys and assessment of breeding performance

For the study area with solitary breeding pairs, the field surveys followed the same methodology as in previous surveys in the same area (Dahlén & Eriksson 2002), and assessments of the breeding performance were based on a sample from a larger population (see above under "study area"). Normally 2–4 visits at each breeding pool were required in order to get enough information for a reliable assessment of the breeding result. At the Tisjökölen site, surveys were made from an observation tower from where the majority of all potential breeding pools could be surveyed.

For both areas, the first visit during each season was done before or during the nesting period, in order to reduce the risk of biased estimates of the breeding performance due to undetected but failed breeding attempts. The timing of the last visit at each site was adjusted in order to follow the survival of chicks until they were classified as *large-sized:* being more than half of the length the adult parent birds. A few breeding records, where it was not possible to acquire the full information to meet these requirements, were excluded in the assessments of breeding performance.

For the two breeding categories, the annual productivity was measured by calculating *the mean number of large-sized chicks per nesting pair,* e.g. that breeding was confirmed by finding a nest with eggs or an incubating bird. Despite the fact that the risk of not detecting breeding attempts that failed before hatching was minimised, it cannot be excluded that some figures may be a slight overestimate. The same approach was however, applied for solitary and colonial breeding pairs during the whole study period. It is thus most likely, that this potential error has not introduced any bias that might affect comparisons between sites and or over time.

Breeding success was assessed by calculating daily survival rates, using the Mayfield method (as presented by Beintema 1996), assuming that an event such as the start of incubation, hatching or failure occurred at the mid-point date between two successive visits to the breeding site. The analyses were split up into assessments of *hatching success,* i.e. the probability that a clutch would hatch, and *brood rearing success*, i.e. the probability that at least one chick would survive to fledge. We assumed an average incubation period of 28 days and a fledging period of 40 days, on the basis of handbook information (e.g. Cramp & Simmons 1977, Barr et al. 2000) as well as unpublished data from voluntary-based surveys of the breeding performance in Sweden (Projekt LOM, http://www.birdlife.se). Broods were considered as lost, if chicks were last seen when smaller than half-grown, but not recorded at later visits at the same site, and fledged if the chicks were larger than half-grown when seen for the last time. This approach includes a risk that broods lost soon after hatching may have been overlooked and thus erroneously classified as a failure late during the incubation stage, resulting in an underestimate of the assessment of hatching success while brood rearing success might have been overestimated. Any impact on the interpretation of the results is elaborated upon in the discussion section.

Finally, we used the annual *mean percentage of broods with two large-sized chicks* to get an indication of the survival of chicks until fledged. Most loon pairs lay two eggs (Dahlén & Eriksson 2002, with reference to the studied population) and the chicks' survival is primarily dependant on the parents' food provisioning success, as has been indicated for the Red-throated Loon (Okill & Wanless 1990, Eberl & Picman 1993, Ball 2004, Rizzolo et al. 2014) as well as for the Black-throated Loon *Gavia arctica* (Jackson 2003). For these calculations we included breeding attempts detected only during the chick rearing stage.

Statistical analyses

As only parts of the data set were normally distributed (as checked with the Shapiro-Wilk test, http://scistatcalc.blogspot.se/2013/10/shapiro-wilk-test-calculator.html), non-parametric alternatives were used for the statistical analyses. Hence, comparisons between solitary and colonial breeding pairs were done by pairing of data for the same year for both categories by using the Wilcoxon's test for matched pairs, and temporal trends in the breeding performance were investigated by calculating the Spearman rank correlation coefficient r_s (e.g. Fowler & Cohen 1995; two-tailed significance levels presented).

Results

Solitary breeding pairs

The average annual productivity was 0.63 (± S.D.0.16) large-sized chicks per pair and year for solitary breeding pairs (details in Figure 3). This

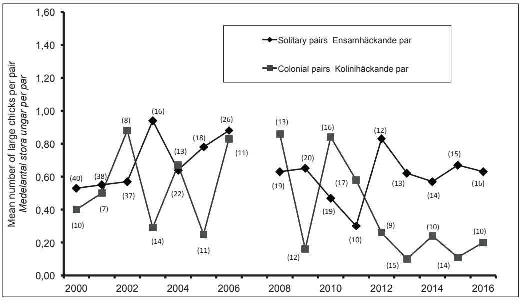


Figure 3. Mean number of large-sized chicks per pairs in solitary breeding and colonial breeding Red-throated Loon *Gavia stellata* pairs. Number of pairs included in the assessments is given in parenthesis.

Medelantal stora ungar per häckande par hos ensamhäckande och kolonihäckande smålommar. Antal par som ingår i beräkningsunderlaget anges i parentes.

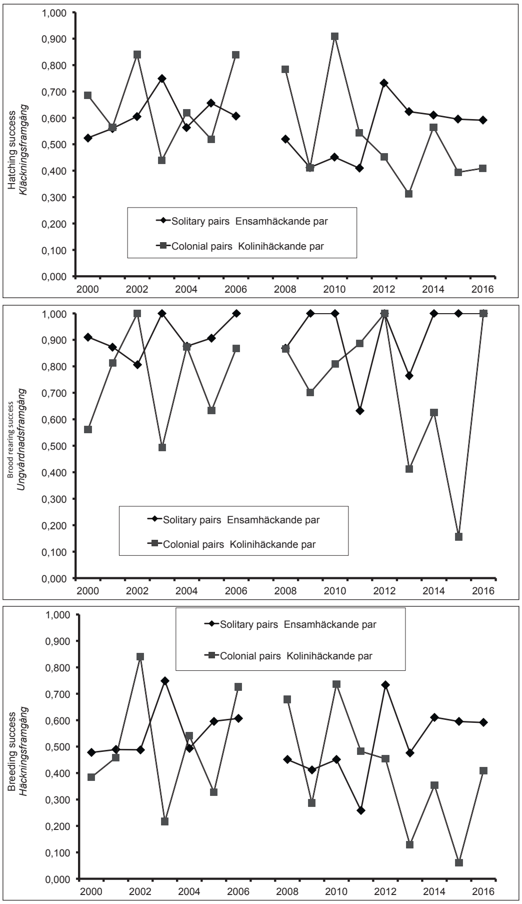


Figure 4. Breeding success in solitary breeding and colonial breeding Red-throated Loons *Gavia stellata.* Detailed data in Table 1.

Häckningsframgång hos ensamhäckande och kolonihäckande smålommar. Detaljdata i tabell 1.

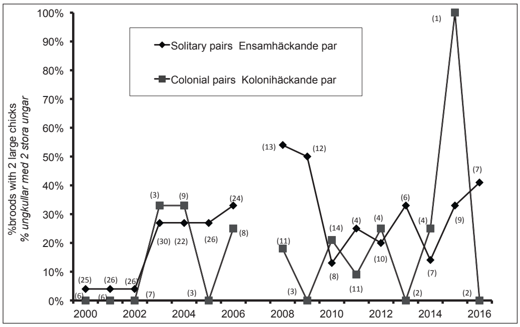


Figure 5. Percentage of broods with two large-sized chicks in solitary breeding and colonial breeding Red-throated Loon *Gavia stellata* pairs. Number of broods included in the assessments is given in parenthesis.

Procentandelen ungkullar med två stora ungar hos ensamhäckande och kolonihäckande smålommar. Antal ungkullar som ingår i beräkningsunderlaget anges i parentes.

figure is at the same level as recorded for South-central Sweden (Svealand and Dalsland) as a whole for the period 1994–2013 (Table 1 in Eriksson 2014). There was therefore no indication of any deviation from the overall breeding performance in a larger geographical context.

The mean hatching probability over years was 0.576 and the average probability that at least one chick in a brood would become fledged was assessed to be 0.915 over years, making an overall probability of successful breeding of 0.530 over years (Figure 4, details in Table 1a). The mean percentage over years of broods with two chicks was 24% (details in Figure 5).

There were no indications of any temporal trends (Spearman r_s = -0.10 for the mean number of chicks per pair, 0.04 for the hatching success, 0.33 for the brood rearing success, 0.07 for the overall breeding success, and 0.30 for the percentage of broods with two chicks, N = 16 years).

Colonial breeding pairs

For colonial breeding pairs the average annual productivity was 0.52 (± S.D. 0.29) large-sized chicks per pair and year (details in Figure 3), and although lower it was not statistically different from the results for solitary breeders (Wilcoxon's test for matched pairs, p = 0.30).

The mean hatching probability over years was 0.581 or the same as for solitary breeding pairs (Wilcoxon's test for matched pairs, p = 1.00) although, there was an indication of decline over the study period (Spearman r_s = -0.52, $0.05<p<0.10$). The average probability that at least one chick be-

Table 1. Primary data for calculations of breeding success using the Mayfield method.
Primärdata för beräkning av häckningsframgången, enligt Mayfield-metoden.

a) Solitary breeding pairs *Ensamhäckande par*

	2000	2001	2002	2003	2004	2005	2006	2008	2009	2010	2011	2012	2013	2014	2015	2016
Number of nests *Antal bon*	28	34	36	16	17	14	20	17	19	17	10	12	13	13	15	12
Number of exposure days *Antal exponeringsdagar*	600	573	719	385	242	264	445	338	311	347	216	269	353	338	374	264
Number of failed clutches *Antal misslyckade häckningar*	14	12	13	4	5	4	8	8	10	10	7	3	6	6	7	5
Daily survival rate *Daglig överlevnad*	0.977	0.980	0.982	0.990	0,980	0.985	0.982	0.977	0.969	0.972	0.969	0.989	0.983	0.983	0.982	0.981
Hatching probability *Sannolikhet för kläckning*	0.524	0.560	0.606	0.749	0.564	0.656	0.607	0.519	0.412	0.451	0.409	0.733	0.624	0.611	0.595	0.591
Number of broods *Antal ungkullar*	26	28	28	26	23	15	21	11	10	7	3	9	7	7	7	7
Number of exposure days *Antal exponeringsdagar*	857	880	924	516	602	411	613	283	242	187	87	218	149	197	216	179
Number of lost broods *Antal förlorade ungkullar*	2	3	5	0	2	1	0	1	0	0	1	0	1	0	0	0
Daily survival rate *Daglig överlevnad*	0.998	0.997	0.995	1.000	0.997	0.998	1.000	0.997	1.000	1.000	0.989	1.000	0.993	1.000.	1.000	1.000
Probability of at least one chick being fledged *Sannolikhet att åtminstone en unge blir flygg*	0.911	0.873	0.806	1.000	0.876	0.907	1.000	0.868	1.000	1.000	0.633	1.000	0.765	1.000.	1.000	1.000
Overall probability of successful breeding *Sammantagen sannolighet för lyckad häckning*	0.478	0.489	0.488	0.749	0.494	0.596	0.607	0.451	0.412	0.451	0.259	0.733	0.477	0.611	0.595	0.591

Table 1. Continued.

b) Colonial breeding pairs *Kolonihäckande par*

	2000	2001	2002	2003	2004	2005	2006	2008	2009	2010	2011	2012	2013	2014	2015	2016
Number of nests *Antal bon*	10	9	8	14	11	10	11	12	12	11	16	9	15	10	12	6
Number of exposure days *Antal exponeringsdagar*	221	97	160	269	232	253	318	343	247	297	318	174	259	243	266	185
Number of failed clutches *Antal misslyckade häckningar*	3	2	1	8	4	6	2	3	8	1	7	5	11	5	9	6
Daily survival rate *Daglig överlevnad*	0.987	0.980	0.994	0.971	0.983	0.977	0.994	0.991	0.969	0.997	0.979	0.972	0.959	0.980	0.967	0.969
Hatching probability *Sannolikhet för kläckning*	0.686	0.565	0.840	0.440	0.620	0.519	0.839	0.784	0.410	0.910	0.544	0.452	0.312	0.565	0.394	0.409
Number of broods *Antal ungkullar*	9	8	7	7	9	4	9	11	4	16	12	4	5	5	5	2
Number of exposure days *Antal exponeringsdagar*	275	193	176	224	296	87	277	278	112	376	332	136	134	85	84	23
Number of lost broods *Antal förlorade ungkullar*	4	1	0	4	1	1	1	1	1	2	1	0	3	1	4	0
Daily survival rate *Daglig överlevnad*	0.986	0.995	1.000	0.983	0.997	0.989	0.996	0.996	0.991	0.995	0.997	1.000	0.978	0.988	0.955	1.000
Probability of at least one chick being fledged *Sannolikhet att åtminstone en unge blir flygg*	0.561	0.813	1.000	0.493	0.874	0.633	0.866	0.866	0.701	0.809	0.887	1.000	0.412	0.626	0.156	1.000
Overall probability of successful breeding *Sammantagen sannolighet för lyckad häckning*	0.385	0.459	0.840	0.217	0.541	0.328	0.726	0.679	0.287	0.736	0.482	0.452	0.129	0.354	0.061	0.409

Red-throated loons in their breeding tarn. Photo: Christer Brostam.
Smålommar i häckningstjärnen. Foto: Christer Brostam.

came fledged was assessed to be 0.731 and significantly lower than that for solitary breeding pairs (Wilcoxon's test for matched pairs, p = 0.01). Nevertheless, the overall probability of successful breeding was 0.443 over years and not significantly lower than for solitary breeding pairs (Wilcoxon's test for matched pairs, p = 0.28, Figure 4, details in Table 1b).

The mean percentage of broods with two chicks over years was 18% (details in Figure 5) and did not differ from the solitary breeding pairs (Wilcoxon's test for matched pairs, p = 0.29).

As for solitary breeding pairs there were no indications of any temporal trends for the mean number of chicks per pair (Spearman r_s = -0.40), for the brood rearing success (Spearman r_s = 0.01), the overall breeding success (Spearman r_s = -0.32) or the percentage of broods with two chicks (Spearman rs = 0.22, N = 16 years).

Fluctuations in breeding performance between years were not synchronized between solitary and colonial breeding pairs (Spearman rs = -0.17 with reference to the mean number of chicks per pair, -0.30 for the hatching success, -0.17 for the brood rearing success, -0.26 for the overall breeding success, and 0.29 for the percentage of broods with two chicks, N = 16 years). There was thus no indication that any external factor influenced the breeding performance in a similar way for both categories.

Discussion

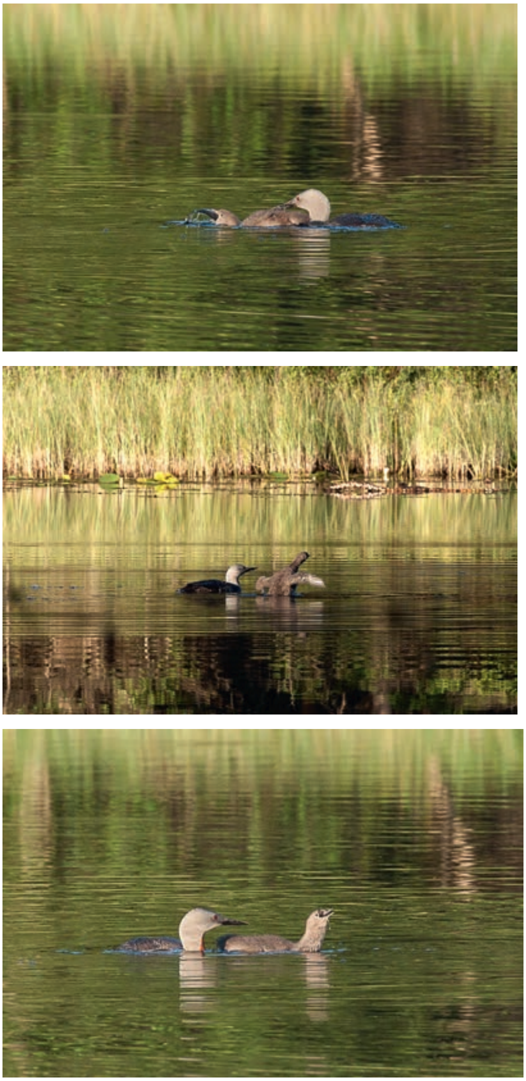
Coloniality has been an attractive field of research, not least as breeding in colonies includes a balance between fitness benefits and costs, and over the years various predictions derived from the theoretical framework have been investigated in field studies and experiments, spanning a large number of species. But results are far from unambiguous or clear-cut, indicating that breeding in colonies may be adaptive for a variety of reasons and that there are not any simple or universal factors behind its evolution (e.g. Danchin & Wagner 1997, Brown & Brown 2001).

For the Red-throated Loon population in this study, the breeding performance was not different between the pairs breeding solitarily and those breeding in the Tisjökölen colony. The average annual productivity was 0.63 and 0.52 large chicks per pair and year respectively, with the reservation that slightly overestimated but still comparable figures cannot be excluded, even if the risk of not detecting breeding attempts that failed before hatching was minimised.

The percentage of broods with two chicks was similar for both categories at 24% and 18%, respectively, which indicates that the survival of chicks until fledged was the same. Assuming that the chicks' survival to a large extent is related to the parents' success in providing them with food, in line with results from previous studies both on Red-throated Loon (Okill & Wanless 1990, Eberl & Picman 1993, Ball 2004, Rizzolo et al. 2014) and Black-throated Loon (Jackson 2003), our results did not indicate any difference between pairs breeding solitarily and colonially with reference to foraging success.

For both solitary and colonial breeding pairs, the probability of a successful breeding outcome was similar at 0.530 and 0.443, respectively. Average hatching probability over years was the same at 0.576 and 0.581, respectively, while the probability that at least one chick would be fledged was significantly lower for pairs breeding colonially at 0.731 compared to 0.915 for solitary breeding pairs. Nevertheless, the overall breeding success did not differ between the two categories and a likely reason for this is that most breeding failures occurred before hatching, as has been shown to apply for the studied population (Dahlén & Eriksson 2002) as well as elsewhere (e.g. Eberl & Picman 1993), and with predation being the main reason. So although there is a risk that chicks lost soon after hatching might have been overlooked, the indication of lower hatching success in comparison to brood rearing success was nevertheless expected with reference to previous results from various studies.

The lower brood rearing success among the colonial breeding pairs can hardly be related to food provision by the parents, as there was no indication of a different percentage of broods with two large-sized chicks. Furthermore, it does not seem as the pairs breeding in the Tisjökölen colony benefited from any lower predation pressure compared with the pairs breeding solitarily. Hatching probability was the same for both categories and previous studies (referred to above) have indicated that predation is the main reason for failures during the incubation period. Rather, it might be speculated if the concentration of breeding pairs and chicks in a limited area attracted predators, and that this might explain the indication of decline in hatching success as well as the lower brood rearing success among the colonial breeding pairs. Indeed, anecdotal information from field visits at the Tisjökölen site indicates that low breeding success in single years could be related to the presence of e.g. Eurasian Eagle Owl *Bubo bubo,* Western Marsh Harrier



Red-throated Loon feeding young in the breeding tarn with a fish that has been collected in a clear-water lake at some distance. Photo: Kristina Eriksson.
Smålom matar unge i häckningstjärnen med fisk som hämtats i klarvattensjö på visst avstånd.

Circus aeruginosus or Red Fox *Vulpes vulpes*. In addition, there are no observations of any cooperative defence against predators by Red-throated Loons breeding at Tisjökölen, while this was recorded at the site for e.g. Northern Lapwing *Vanellus vanellus* and Mew Gull *Larus canus* against Red Fox. This is in line with previous findings suggesting that coloniality may attract predators rather than reduce the risks of predation (e.g. Varela et al. 2007).

For the Red-throated Loon, and bearing in mind that rigorous field experiments designed in order to test any hypotheses from current theory are lacking, it seems that habitat requirements *per se* determined the selection of breeding sites. Both kinds of habitats required for breeding, e.g. small pools with quagmire shores and freshwater lakes harbouring prey fish, such as small-sized cyprinid or salmonid fish are common elements in the coniferous forest landscape of Scandinavia, but a combination of both at a distance short enough for transporting fish to the chicks may be less common. Therefore, individual pairs do not avoid nesting with conspecifics where a concentration of potential nest-sites is located within a limited area and with profitable foraging waters at a close distance, if the expected breeding outcome is neutral in comparison to solitary breeding at a small and isolated pond. Thus, colonial breeding in the Red-throated Loon is most likely a result of commodity selection.

The result of this study is consistent with what has been found to apply for Red-necked Grebe *Podiceps grisegena*, another water-bird species that exhibits both solitary and colonial breeding. For this species, coloniality has been shown to be linked to the availability of quite rare high-quality habitats rather than any benefits of breeding in close vicinity to conspecifics (Sachs et al. 2007).

Acknowledgments

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Sammanfattning

Kolonihäckning brukar definieras som en ansamling av bon inom ett begränsat område, i kombination med att de viktigaste födokällorna ligger utanför reviret. Företeelsen har varit föremål för ett mycket stort antal undersökningar och har innefattat många olika arter, men det har visat sig vara svårt att hitta några generella förklaringar. Inom ramen för beteendeekologins framväxt under 1970-talet utvecklades ett teoribygge kring tänkbara för- och nackdelar och med fokus på risker för predation och förutsättningarna att hitta föda.

Genom att häcka i kolonier kan vaksamheten mot predatorer bli mer effektiv - flera par ögon ser mer än två. Visserligen kan en koloni attrahera predatorer, men om predationsrisken inte ökar i proportion till kolonins storlek uppstår en utspädningseffekt och risken för varje enskilt bo kan antas minska. Kolonier har också föreslagits fungera som informationscentra; genom att observera och följa sina artfränder kan födosöket förmodas bli mer effektivt för egen del. Men över åren har den teoretiska ansatsen skiftat fokus från individuella fördelar till att tolka kolonihäckande i termer av en mer passiv effekt eller biprodukt av t.ex. habitatval eller sexuell selektion. Sammantaget har kolonihäckning bedömts vara vanligast bland vattenlevande fågelarter och kopplad till bl.a. avsaknad av ett födosöksrevir och hög predationsrisk.

Smålommen passar in i mönstret såtillvida att den är knuten till vattenmiljöer, födosöket sker utanför häckningsreviret och bopredation är en viktig orsak till misslyckade häckningar. Vanligast är att enstaka par häckar vid småsjöar och tjärnar som ofta saknar fisk. Bytesfisk till ungarna hämtas i större klarvattenssjöar eller i havet upptill ett avstånd på ungefär 10 km, sällan längre, och med preferens för fiskevatten med småvuxna bestånd av mört- eller laxartad fisk. Men det förekommer också att flera par ibland häckar tillsammans i koloniartade ansamlingar.

I den här studien jämförs häckningsutfallet hos ensamhäckande smålommar i ett ungefär 1820 km^2 stort undersökningsområde omfattande ungefär 70 par i Malung-Sälens kommun i Dalarna med en koloni belägen på Tisjökölen (Figur 2) som är ett ungefär 0,85 km^2 stort myrområde med 20-talet smålomspar. Oss veterligt har ingen liknande jämförelse gjorts tidigare och med avseende på smålommen.

Metodik

Vad gäller de ensamhäckande smålomsparen bedömdes häckningsutfallet genom att tänkbara häckningsplatser besöktes vid 2-4 tillfällen under häckningssäsongen för att få tillräckligt underlag för att bedöma häckningsresultatet. Vid Tisjökölen kunde huvuddelen av tänkbara häckningsgölar i myrområdet observeras från ett fågeltorn.

Häckningsutfallet bedömdes genom att beräkna *medelantalet stora ungar per häckande par*. Med ”stora” ungar avsågs ungar som blivit minst halvvuxna. Vidare bedömdes *häckningsframgången* med den s.k. Mayfield-metoden, med separata beräkningar av *kläckningsframgången* (= sannolikheten för kläckning) och *ungvårdnadsframgången* (= sannolikheten att åtminstone en unge blir flygg). Slutligen användes *procentandelen ungkullar med två stora ungar* som en indikation på ungarnas överlevnad tills de blivit flygga och under antagandet att ungarnas överlevnad primärt är beroende på hur föräldrafåglarna lyckas med att förse ungarna med fiskbyten (vilket i tidigare undersökningar har visat sig gälla för både smålommen och storlommen).

Resultat

På basis av data från 16 år, 2000-2016 (med undantag för 2007), kan resultaten sammanfattas, som följer:

- Ungproduktionen var i medeltal 0,63 stora ungar per par och år för ensamhäckande par och 0,52 för kolonihäckande par (detaljer i Figur 3, skillnaden är inte statistiskt signifikant). Resultatet ligger på samma nivå som för Svealand (och Dalsland) i övrigt.
- Sannolikheten för kläckning var densamma för ensamhäckande och kolonihäckande par, i medeltal 0,576 respektive 0,581 per år, medan ungvårdnadsframgången var högre för ensamhäckare jämfört med kolonihäckande par, i medeltal 0,915 respektive 0,731 per år. Men den sammanräknade häckningsframgången låg på ungefär samma nivå, i medeltal 0,530 respektive 0,443 per år för ensamhäckande och kolonihäckande par (Figur 4, detaljer i Tabell 1).
- Procentandelen ungkullar med två stora ungar var likartad, i medeltal 24% per år för ensamhäckande par och 18% för kolonihäckande (detaljer i Figur 5).

Det fanns en indikation på att häckningsframgången för kolonihäckade par försämrats, men därutöver noterades inga tidstrender i häckningsutfallet, vare sig för ensamhäckande eller kolonihäckande par. Variationerna i häckningsutfallet var inte synkroniserade, så det fanns ingen indikation på att externa faktorer har påverkat ensam- och kolonihäckande par på ett likartat sätt.

Diskussion

Det fanns inga indikationer på skillnader i häckningsutfallet eller att kolonihäckning skulle vara kopplad till någon fördel som kan mätas i termer av ungproduktion. Det faktum att procentandelen ungkullar med två stora ungar var densamma antyder att det inte var någon skillnad i ungarnas överlevnad tills de blivit flygga och således inte heller vad gäller föräldrafåglarnas möjligheter att finna bytesfisk till ungarna. Det är alltså inte troligt att kolonihäckningen har innefattat någon fördel med hänsyn till förutsättningarna för födosöket.

Sannolikheten för ett lyckat häckningsutfall var likartad och ungefär hälften av häckningarna lyckades i den meningen att åtminstone en unge blev flygg. Kläckningsframgången var lägre än ungvårdnadsframgången och nästan identiskt lika i de båda grupperna. Även i tidigare undersökningar har man noterat att de flesta av de misslyckade häckningarna överges under ruvningsstadiet och att bopredation är en av de viktigaste orsakerna. Det fanns alltså ingen indikation på kolonihäckningen inneburit någon fördel i form av mindre predationsrisk. Snarare kan frågan lämnas öppen om ansamlingen av flera par inom ett begränsat område kan attrahera predatorer, såsom närvaro av t.ex. berguv, brun kärrhök eller räv enstaka år, och att detta kan ha bidraget till den lägre ungvårdnadsframgången bland kolonihäckande par. Det finns inte några observationer av att smålommar gemensamt har försvarat sig mot predatorer, medan detta har noterats både hos tofsvipa och hos fiskmås när en räv befunnit sig ute på myren.

Våra resultat pekar på att det torde vara häckningsmiljön i sig, snarare än någon specifik fördel av att häcka i en koloni tillsammans med artfränder som förklarar varför smålommen ibland häckar i koncentrationer med flera par inom ett begränsat område. De två miljöer som smålommen behöver för sin häckning, dvs. småsjöar eller tjärnar med gungflyartade stränder med lämpliga boplatser och klarvattenssjöar med goda bestånd av småvuxen lax- eller mörtartad fisk är var för sig ganska vanliga i det skandinaviska barrskogslandskapet. Men det är kanske mindre vanligt att de båda miljöerna förekommer tillsammans med inbördes avstånd som inte är för stora för att försvåra transporten av bytesfisk till ungarna. En gynnsam kombination av flera lämpliga boplatser inom ett begränsat område och med lämpliga fiskevatten på nära avstånd kan alltså vara anledning till att flera smålomspar ibland häckar tillsammans, under förutsättning att det förväntade häckningsutfallet är ungefär detsamma som för ensamhäckande par. Resultatet ligger i linje med vad man funnit för gråhakedoppingen, en annan vattenlevande fågelart där det finns både ensamhäckare och kolonihäckare, och där kolonihäckning kan förklaras genom en gynnsam kombination av olika miljöfaktorer.

Tack

Undersökningen har genomförts inom ramen för Projekt LOM, som på frivilligbasis utför ett rikstäckande fältarbete som syftar till att övervaka storlommens och smålommens häckning. Över åren har ekonomiska bidrag bland annat erhållits från bland annat Alvins Fond och Världsnaturfonden-WWF. Två granskare har bidragit med synpunkter och kommentarer på en tidigare textversion.

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Effects of harbour and ship channel construction on breeding seabirds

Effekter av hamn- och kanalkonstruktion på häckande sjöfåglar

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Abstract

Bird population changes were studied during the construction of the new Vuosaari Harbour in 2001–2011 (Helsinki, Finland, N60°14.52’ E25°9.07’). Part of the monitoring programme was to determine the possible impacts of harbour and ship channel construction on breeding seabird populations. The study revealed that the population trends of most species showed no substantial difference between islets close to harbor and channel and more distant reference islet. A Black-headed Gull colony was destroyed by landfill but that was deliberately and with permission. The trends of Ruddy Turnstone and Great Black-backed Gull populations were more negative in near than in distant islets. In the surrounding archipelago, the Barnacle Goose, Mew Gull and Common Black-headed Gull clearly increased during the study years. The European Herring Gull decreased distinctly, which was probably caused by the removal of individuals from the rubbish dumps around Helsinki. There were some differences between the restricted- and open-access islands. The population of Herring Gull was higher and the decline stronger in the restricted areas, while the Common Ringed Plover population simultaneously increased.

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Introduction

The seabird populations of the northern Baltic Sea are threatened by many different factors. Boating and other disturbances such as fishing, eutrophication, feral mink, birds of prey, weather, toxic substances and many other factors have been mentioned as the probable causes of poor fledgling production and seabird population declines (Hario & Uuksulainen 1993, Hario et al. 1987, Nordström et al. 2003, Hario 2004, Hario, Mazerolle, Saurola 2009, Skov et al. 2011, Hario & Rintala 2011, 2014). Dredging, dumping and harbour construction probably bring about negative impacts on seabirds breeding at sites near construction areas.

Most studies of the effects of human activity impacts on the environment have been conducted in Europe and North America. Environmental Impact Assessments (EIAs) were first formally established in the USA in 1969, and in 1985 the European Community directive on EIAs was introduced (Glasson et al. 1994, 2012). During recent decades, the ecological effects of urbanization and other human activities have been widely studied, and several reports have been compiled on the subject (McDonnell et al. 2009, Niemelä 2011). The sphere of influence of urbanization and the direction and size of its effect on animals may vary, depending on the habitat type and the spatiotemporal scale of human activities. Some species are also more sensitive than others. The negative effects of urbanization, e.g. on birds, may extend to areas as far as several kilometres away from the source of the disturbance (Watts & Bradshaw 1994, Kala- ja Vesitutkimus Oy et al. 1996, Rodgers & Smith 1997, Mensing et al. 1998). Even small variations from the human standpoint in distance between the disturbance source and birds or in vessel speed can translate into remarkable variation in the disturbance effect. For example, the disturbance of boat traffic on foraging Black Guillemots Cepphus grylle was studied at a breeding colony in the Bay of Fundy, Canada, showing that the species was susceptible to disturbance by approaching vessels. The investigators concluded that guillemot flushing probability would be reduced by 10% any time on condition that a setback distance of at least 600 m from the shore was established and maximum speed limited to 25 km/h (Ronconi & St. Clair 2002). In another study, a general setback distance of 50 m between boats and birds was estimated to be sufficient to prevent disturbances to most seabird nesting and roosting sites while allowing viewers to appreciate the seabirds; at shorter distances, the disturbance effect on birds was clearly strengthened (Chatwin et al. 2013).

The planning of a new harbour in Helsinki was initiated as early as the mid-1960s, with the aim of starting the Vuosaari Harbour project in 1992. The EIA procedure of the project was initiated in 1994, when the new environmental act came into force. The town council of Helsinki accepted the establishment plan of the harbour in 1996 (Heikkonen 2008).

The harbour was located next to a Natura 2000 area (F10100065, 'Mustavuoren lehto ja Östersundomin lintuvedet'). Before the start of the project, environmental authorities and nature protection organizations argued, that construction of a new harbour and road connections could seriously affect breeding bird populations. They also argued that enough knowledge was not available, and that construction was not permissible according to so called precautionary principle.

The effects of harbour construction on the Natura values were investigated based on various monitoring programmes. The objective of these programmes was to prevent possible harmful environmental effects and to document the impacts observed and methodology used. The monitoring programmes were carried out as a cooperative project between the Helsinki Environment Centre and the Port of Helsinki. One part of the operation included the monitoring of birds, which began in the archipelago in 2001 and on the land areas in 2002 and continued to as recently as 2011 (Koskimies 2001).

About 30–40 species of seabirds (ducks, geese, waders, gulls and divers) are found in the Gulf of Finland, which are common breeders or migrants along shoreline areas. The most abundant breeding seabird species in the sea area are the Common Eider *Somateria mollissima*, Black-headed Gull *Larus ridibundus*, Mew Gull *Larus canus*, Arctic Tern *Sterna paradisaea* and European Herring Gull *Larus argentatus* (Hario & Rintala 2011).

The populations of many seabird species have exhibited remarkable long-term trends at large spatial scales, and for many species the mechanisms driving these trends are often unknown. In addition to natural reasons, the causes may be related to human activities. During recent decades, the Barnacle Goose *Branta leucopsis*, Canada Goose *Branta canadensis* and Great Cormorant *Phalacrocorax carbo* numbers have increased rapidly, especially in the western part of the Gulf of Finland (Hario & Rintala 2011). These species have benefited from various human activities with the consequence of developing new foraging and resting areas, and larger fish stocks, resulting from eutrophication.

The purpose of this study was to investigate the effects of the Vuosaari Harbour and ship channel construction on the local seabird populations. We aimed at revealing possible changes in the archipelago bird populations near the construction area and at pointing out the factors that had altered the population trends. In particular, our study concerned the population trends of rare or vulnerable species. From the environmental management standpoint, the focus was on whether construction had such negative effects on birds that could be avoided or mitigated during similar construction processes. During the monitoring project, the port and environmental authorities were informed regularly about the annual results from the field.

Materials and methods

The seabird populations were monitored, using yearly breeding bird counts. The monitoring continued for three years after opening of the Vuosaari Harbour in November 2008. The possible effects on the seabird populations were investigated in two areas. One area included islets within two kilometers from the harbor and ship channel. Another area included islets at a larger distance than two kilometers. The birds in the former area were considered to be at risk of disturbance and we call this area the risk area. We assumed that two kilometers was sufficient to exclude disturbance and this area is our reference area.

Study areas

The risk area comprised 17 islets (Figure 1). Two islets, Varisluoto and Västinki, were covered by harbour landfills and the construction area in 2004,

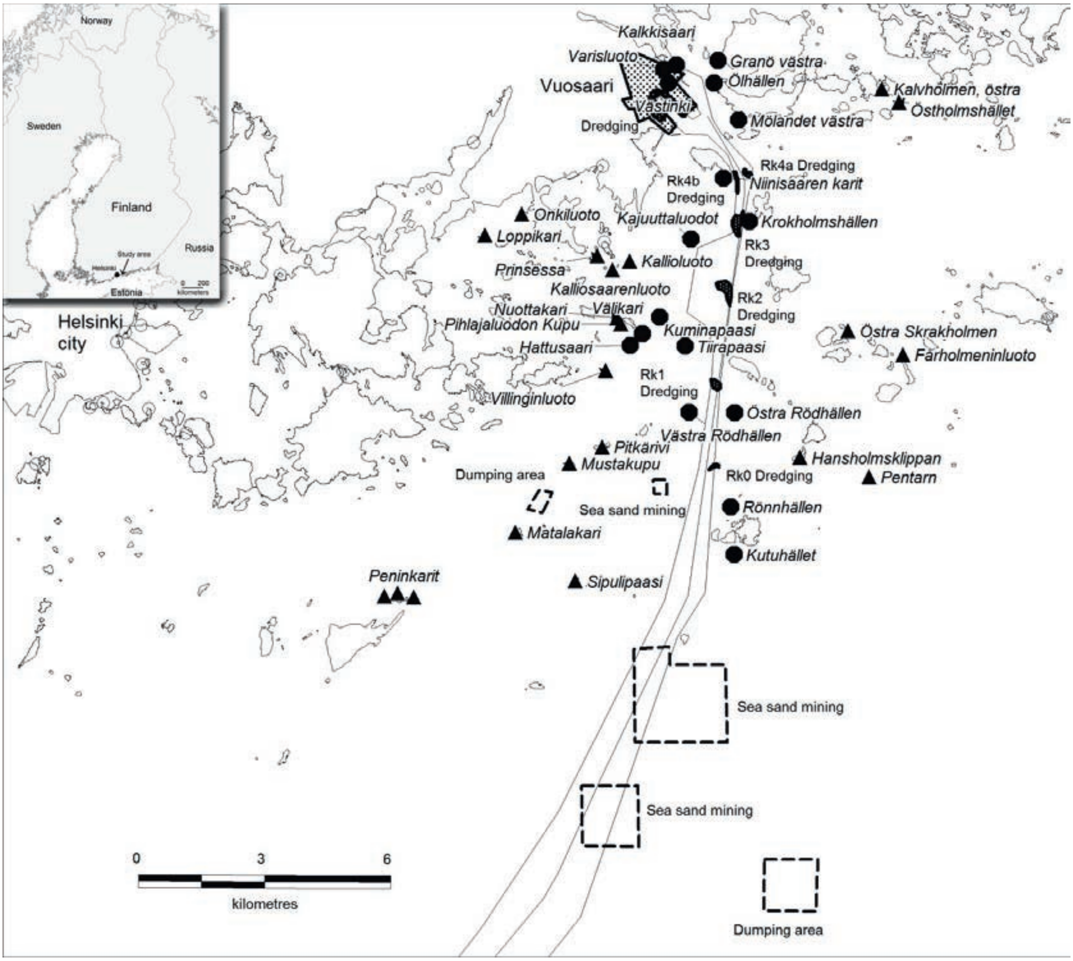


Figure 1 The study area. The risk islets are marked with black circles and reference islets with triangles. Dredging, dumping and mining areas are also shown.
Undersökningsområdet. Öar med störningsrisk markerade med svart cirkel och referensöar med triangel. Område för schaktning och dumpning markeras också.

with permission of the authorities. Västinki, an islet with a colony of Black-headed Gulls (200 pairs), was covered with landfills after the breeding season, preventing breeding in subsequent years. All except one risk islet were open-access or under recreational use. Usually, in the archipelagoes between the communes of Sipoo and Espoo, the nesting success of seabirds has been poorer on open islets subject to recreational use than on islets in the conservation or military areas (Matti Luostarinen, personal observations).

The reference area included 21 islets (Figure 1). They were selected, based on their seabird fauna, to be similar to the risk islets at the same distance from the harbour. Open-access and recreational use were allowed for 12 islets. Nine islets had restricted access: three were military areas and the remaining six were nature reserves.

The islets in the archipelago differ: the inner archipelago islets are more luxuriant, and their shores are usually less susceptible to eroding surge. In the outer archipelago, the islets are more rugged. Hence, the seabird communities also differed along the inner/outer archipelago aspect.

Seabird population counts

The seabird counts of the monitored islets were based on the bird monitoring programme in Koskimies (2001) that was part of the larger monitoring programme on the impacts of the Vuosaari Harbour project. Matti Luostarinen performed all the counts during the years 2001–2011.

We aimed at visiting each islet three or four times during the breeding season. In May, the early breeders were counted. In June, the numbers of late breeders were registered, and the adult birds of these species and the young of the early-nesting species were ringed. In late June to early July, the young of the late-nesting species were ringed.

During several survey years, bad weather occasionally prevented fieldwork, and some islets were visited fewer than three times during a season. Counts were taken during a 15–30-min stay per islet; a longer census time would have resulted in damage to the nesting sites. In some locations, the terrain was so difficult that the time-consuming nest censuses were not done so as to minimize disturbance; in these cases, we quickly performed a rough population estimate, based on the number of adult birds in the vicinity of the breeding colonies. On the breeding islets of the Caspian Tern *Hydroprogne caspia*, the counts were performed with special care and rapidly.

We combined the numbers of Common Tern *Sterna hirundo* and Arctic Tern, because at some mixed-breeding sites the periods required for separating all adult birds would have been too lengthy, and only the number of 'terns' was counted. Note that the number of reference islets was higher, and thus the total population of terns on these reference islets could have been larger despite the lower mean number of pairs.

Statistical analyses

The purpose of testing was to reveal possible differences in population size or trends between the risk and reference islets. For the statistical analyses, the area factor classified each islet as (1) a risk or (2) a reference islet, and the use-of-islet factor classified the islets as either (1) open-access or (2) restricted-access, due to their protected or military use status.

To estimate the differences in species-specific population trends by risk/reference and open-access/restricted areas, each species having a total number of more than 30 pairs was analysed with generalized additive mixed models (GAMMs), primarily for denoting the smooth term (Wood 2004, Zuur et al. 2009) of the trends by the spatial treatment (i.e. the trends between the risk and reference islets) and generalized linear mixed models (GLMMs) for revealing the interactions of the year-effect and the spatial treatment and island type (i.e. open-access vs. restricted-access). The islet or island identification code was set as a random effect, and the first-order residual temporal autocorrelation structure was controlled in all models. The GAMMs were performed, using the function gamm of library mgcv (mixed GAM (generalized additive model) computation vehicle) and the GLMMs with the function glmmPQL (glmm Program Query Language) of library MASS (Modern Applied Statistics with S) (Venables & Ripley 2002) in R vers. 3.1.2 (R Core Team 2014). Passerine species were not tested, because on many islets the population data of these species were only of the present/absent type. The Black Guillemot bred only on the reference islets.

Results

In all, 28 seabird species were observed breeding on the risk islets during the study period in 2001–2011, and the annual total pair number of these varied from 980 to 1196. For the reference islets, the corresponding numbers were 32 species and pair numbers varying from 1090 to 1615 (Appendix 1).

Species-specific models revealed several significant parameter effects on the overall trends and deviations in the trends between the risk and reference areas, as well as inter-area variation in islet-specific pair densities (Table 1, Figure 2). The Ruddy Turnstone *Arenaria interpres* trend was somewhat more negative in the risk than in the reference area, as was the trend for the Great Black-backed Gull *Larus marinus*.

The pair numbers of the Mute Swan *Cygnus olor* decreased generally, showing lower densities on the reference islets. The Canada Goose numbers increased significantly on the reference islets, but not on the risk islets. The numbers of Barnacle Goose increased slightly faster on the reference than on the risk islets.

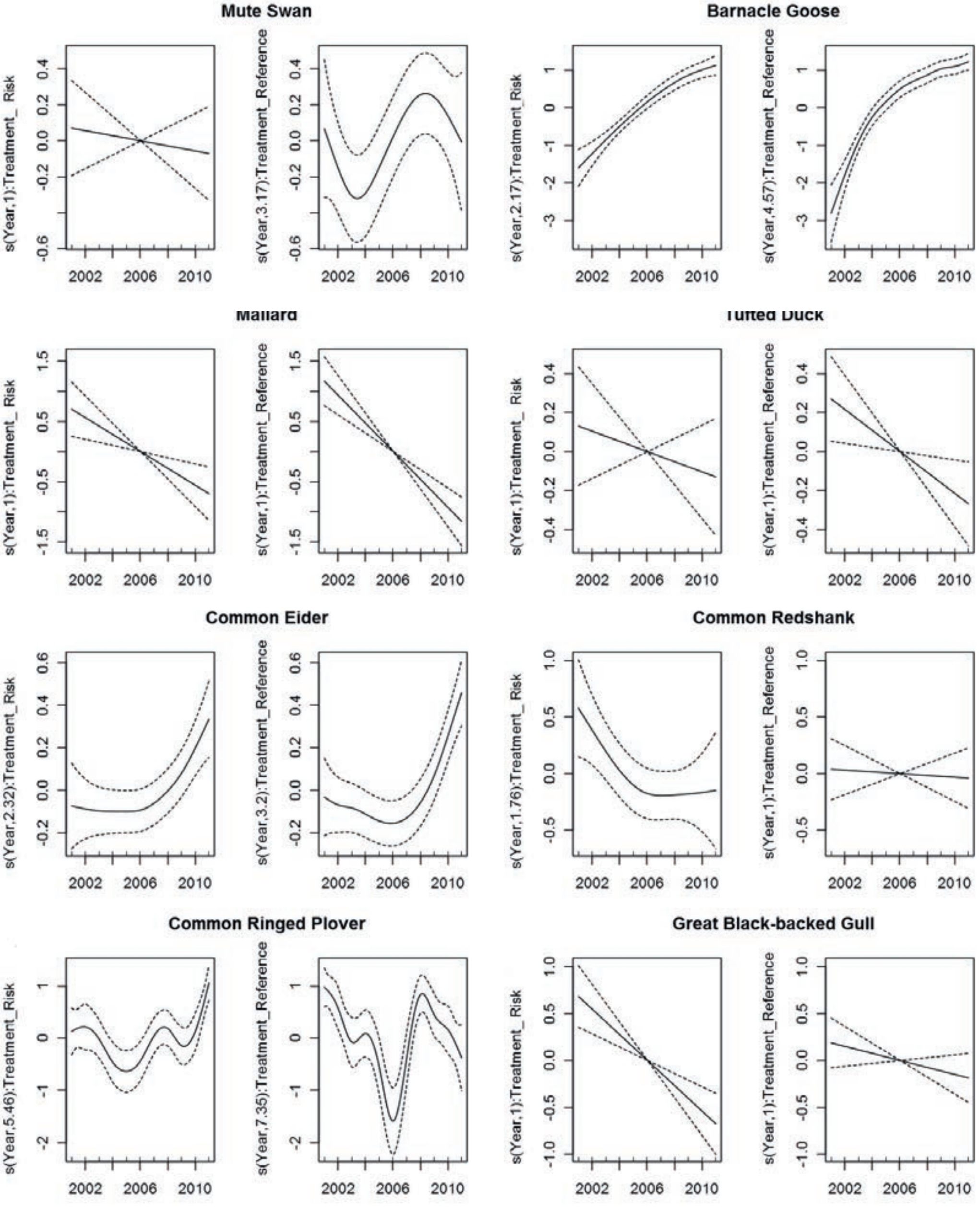
The Mallard *Anas platyrhynchos* declined significantly on both the risk and reference islets, showing significantly higher pair densities on the reference islets. The Tufted Duck *Aythya fuligula* declined significantly, particularly on the reference islets. The numbers of Common Eider showed exponential increases on both the reference and risk islets during recent years.

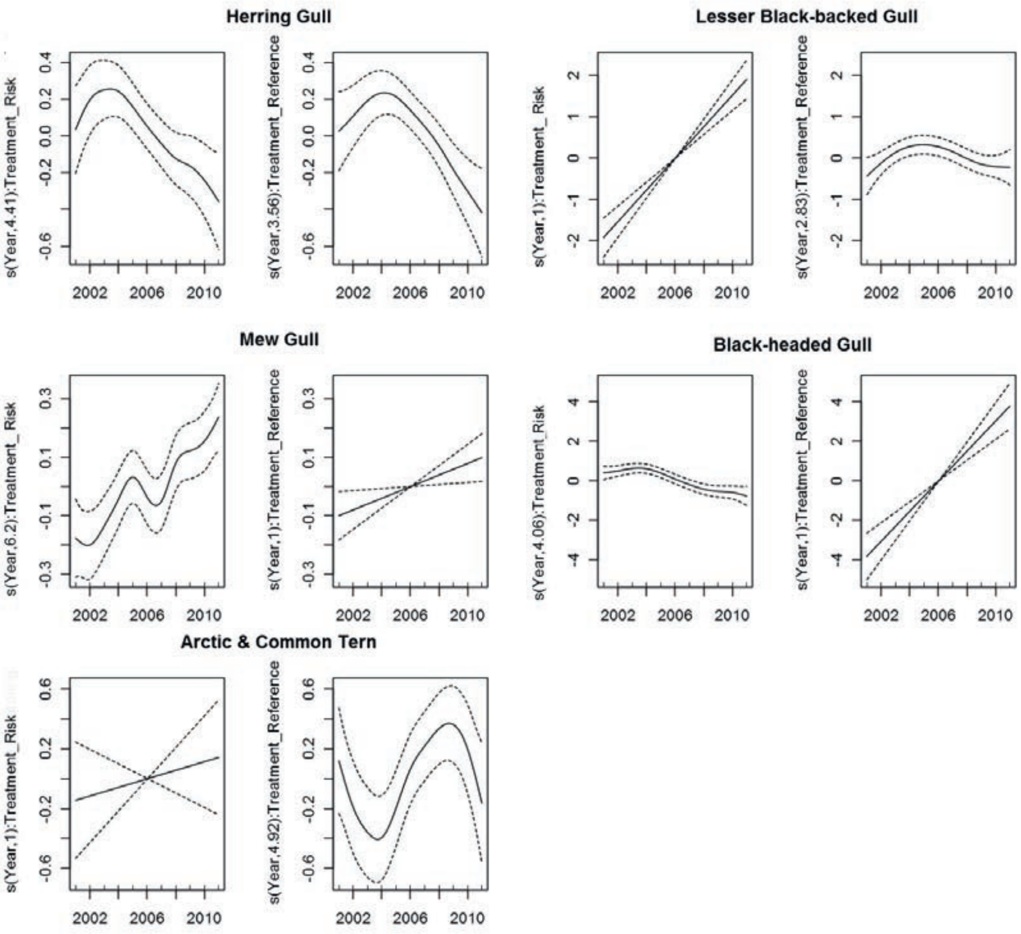
The Eurasian Oystercatcher *Haematopus ostralegus* showed general increases on both the risk and reference islets. The numbers of Common Redshank *Tringa totanus* decreased slightly, particularly on the risk islets at the beginning of the study

Table 1 Statistically significant parameters (Term) affecting seabird numbers, based on two model types (m.typ): generalized additive mixed models (GAMM) and generalized linear mixed models (GLMM). The coefficient is the parameter value indicating the direction and strength of the effect, with its standard errors (Std.Er), degrees of freedom (DF), t-statistics (t) and statistical significance (p).
Statistiskt signifikanta parametrar (Term) som påverkar antal sjöfåglar enligt två modeller (m.type). Coefficienten anger riktning och styrka för effekten (med standardfel, frihetsgrader, t-värde och signifikans).

Species	Term	Coefficient	Std.Error	DF	t	p	m.type
Canada Goose	Year:Reference area	0.223941	0.097676	368	2.292687	0.0224	GAMM
Barnacle Goose	Year:Risk area	0.789187	0.218527	368	3.611395	0.0003	GAMM
	Year:Reference area	1.887489	0.465917	368	4.051128	0.0001	GAMM
	Year	0.2425	0.02735	368	8.865754	0.0000	GLMM
Mute Swan	Year	-0.01056	0.00116	366	-9.07058	0.0000	GLMM
	Reference area	-193.182	20.58107	34	-9.38638	0.0000	GLMM
	Year:Reference area	0.09587	0.01024	366	9.360752	0.0000	GLMM
Mallard	Year:Risk area	-0.44206	0.14119	368	-3.13096	0.0019	GAMM
	Year:Reference area	-0.73385	0.127865	368	-5.73929	0.0000	GAMM
	Year	-0.1369	0.0000	366	-119.584	0.0000	GLMM
	Reference area	151.9121	3.0000	34	50.85635	0.0000	GLMM
	Year:Reference area	-0.0759	0.0000	366	-59.4027	0.0000	GLMM
Tufted Duck	Year:Reference area	-0.1692	0.068136	368	-2.48329	0.0135	GAMM
	Year	-0.04082	0.01606	368	-2.54092	0.0115	GLMM
Common Eider	Year	0.04226	0.0165	366	2.556274	0.0110	GLMM
Oystercatcher	Year	0.04525	0.02045	368	2.212871	0.0275	GLMM
Ringed Plover	Year:Risk area	1.908575	0.779237	368	2.449288	0.0148	GAMM
	Year	0.123	0.03622	366	3.395585	0.0008	GLMM
	Reference area	649.9509	119.2349	34	5.451011	0.0000	GLMM
	Restricted area	336.2443	152.4899	34	2.205026	0.0343	GLMM
	Year:Reference area	-0.3239	0.05946	366	-5.4478	0.0000	GLMM
	Year:Resticted area	-0.1656	0.07601	366	-2.17872	0.0300	GLMM
	Reference area:Restricted area	-872.391	196.9603	34	-4.42927	0.0001	GLMM
	Year:Reference area: Restricted area	0.4329	0.09819	366	4.408679	0.0000	GLMM
Redshank	Year	-0.08144	0.0000	366	-2.05288	0.0408	GLMM
Ruddy Turnstone	Year:Risk area	-0.34523	0.119338	368	-2.8929	0.0040	GAMM
	Year	-0.11182	0.03801	368	-2.94229	0.0035	GLMM
Mew Gull	Year:Reference area	0.062564	0.025878	368	2.417683	0.0161	GAMM
	Year	0.04179	0.00922	366	4.531292	0.0000	GLMM
Great Black-backed Gull	Year:Risk area	-0.42825	0.103255	368	-4.14753	0.0000	GAMM
Herring Gull	Year	-0.03796	0.01881	368	-2.0176	0.0444	GLMM
	Restricted area	204.1474	47.46774	35	4.300761	0.0001	GLMM
	Year:Resticted area	-0.10177	0.02367	368	-4.29891	0.0000	GLMM
Lesser Black-backed Gull	Year:Risk area	1.203814	0.147891	368	8.139853	0.0000	GAMM
	Year	0.3916	0.0000	366	8.782242	0.0000	GLMM
	Reference area	837.9104	90.0000	34	9.327097	0.0000	GLMM
	Year:Reference area	-0.4164	0.0000	366	-9.3137	0.0000	GLMM
Black-headed Gull	Year:Reference area	2.40681	0.370284	368	6.499902	0.0000	GAMM
Common and Arctic Tern total	Year:Reference area	-1.1795	0.355281	368	-3.3199	0.0010	GAMM

Figure 2. Modelled population changes of some seabird species on the risk and reference islets.
Modellerade beståndsförändringar för vissa sjöfåglar på risköar och referensöar.





period. The Common Ringed Plover *Charadrius hiaticula* densities were generally higher on the reference than on the risk islets. In the risk area, the densities were generally highest on the restricted islets. However, in the reference area, the densities were lowest but temporally increased most quickly on the restricted islets (Table 1).

Our monitoring data revealed the most substantial changes in numbers for highly colonial species such as the gulls (Table 1, Figure 2). The European Herring Gull population showed a general decline, with highest densities on the restricted-access islets. The Lesser Black-backed Gull *Larus fuscus* increased significantly on the risk islets, while the Mew Gull increased, particularly on the reference islets, but also in the risk area. At the beginning of the study period, the Black-headed Gull was more numerous on the risk than on the reference islets. However, during and after the harbour construction, the pair numbers increased significantly in the reference area.

The numbers of the Common Tern and Arctic Tern fluctuated substantially during the study period (Figure 2), but the modelling indicated no clear associations with the environmental change. The nonlinear model suggested a population decrease on the reference islets.

Discussion

The Vuosaari Harbour and ship channel construction represent one aspect of the urbanization process in the City of Helsinki, which has been intensifying and expanding during recent decades. The number of people in the region of Helsinki is increasing, and the recreation pressure on the archipelago area is strong. The influence of urbanization on bird populations varies, depending on the requirements of a species, but previous observations have shown that construction activity can easily result in negative impacts. The behaviour of gulls and ducks was monitored during a certain new construction activity in Helsinki, which indicated that these seabirds were frightened by the noise at a distances of hundreds of metres (Kala- ja Vesitutkimus Oy et al. 1996). Similar results were also reported with Great Blue Heron Ardea herodias and foraging waterfowl species from Chesapeake Bay and Florida in the USA (Watts & Bradshaw 1994, Rodgers & Smith 1997).

Despite of human activities, the populations of some seabird species have quite uniform trends along wide coastal areas of Finland. For example, the population size developments of Canada Goose, Barnacle Goose, Tufted Duck, Common Redshank, Ruddy Turnstone and European Herring Gull in the Gulf of Finland and other sea areas of southern Finland showed quite synchronous patterns compared with our results.

In our study area the Mallard and Tufted Duck declined during the study years. A similar development has been going on in the longer period1986–2013, when the Tufted Duck decreased moderately in Finland (Hario & Rintala 2014). The Mallard has increased slightly in inland lake areas (Lehikoinen et al. 2013, Pöysä et al. 2013), but in the archipelago, the species has apparently shown no general population trends (Hario & Rintala 2014).

The Canada Goose and particularly the Barnacle Goose have increased along the coastal Gulf of Finland (Hario & Rintala 2014). Based on our results, the Canada Goose population increased during the very beginning of the study period, but decreased slightly during recent years. The Barnacle Goose population increased strongly in our study area, and this species has adapted very well in breeding in Helsinki city. They breed in the archipelago, but use fields, golf courses or even city parks as feeding areas during summer and autumn. The first pair bred in Helsinki in 1989, and in 2009 the population was estimated at 900 breeding pairs and more than 8000 staging individuals in autumn (Väänänen et al. 2010).

Human has had a strong influence on population changes of large gull species. In the early 20th century, only a few European Herring Gull pairs were breeding in the archipelago southwest of Helsinki. In the early 1980s, the population increased to about 6500 pairs. This population expansion may have been due to the easy availability of edible refuse in rubbish dumps. At the same time, the Great Black-backed Gull population also increased (Bergman 1982), and single breeding pairs still breed in the area. The increase in the European Herring Gull population has resulted in various problems. The number of individuals has increased in rubbish dumps and throughout Helsinki. Some individuals became specialized in preying on Common Eider ducklings or the chicks of smaller gull species. Such behaviour is typical for the Great Black-backed Gull (Bergman 1982). Together with other problems, predation by European Herring Gulls was partly responsible for the decline in the Lesser Black-backed Gull (Hario 1990). During 2004–2007, a special project was targeted at reducing the number of European Herring Gulls around Helsinki. In total, about 15 000 European Herring Gulls and 600 Great Black-backed Gulls were culled at four large open-plan refuse dumps in Uusimaa Province. The aim of the project was to minimize the nuisance effects on humans caused by dense flocks of gull individuals in cities and to reduce the predation pressure on Lesser Black-backed Gull chicks (Hario, Rintala, Tanner 2009). The culling project succeeded fairly well and the European Herring Gull population decreased rapidly after initiation of the project. This was most probably why the European Herring Gull population also decreased in our study area.

In comparison to the large gull species, the populations of the Mew Gull, Lesser Black-backed Gull and Black-headed Gull have increased in our study area. This may have resulted from the presence of fewer predatory European Herring Gull and Great Black-backed Gull individuals after the culling project. Construction of the Vuosaari Harbour and landfills on the Varisluoto and Västinki islets (Figure 1) during 2003–2004 reduced the number of Black-headed Gulls. However, when all the islets were examined, the total pair numbers of Black-headed Gulls returned to the levels observed in the early 2000s, i.e. more than 250 pairs. At the beginning of the study period, there were no Black-headed Gulls on the reference islets. During the last few study years, the numbers on the reference islets increased rapidly, which may have been due to individuals moving from the Västinki islet; the immigration of new individuals from elsewhere is another possibility. Nationwide, the Black-headed Gull declined in the early 2000s, but later its population began again to increase (Hario & Rintala 2014).

Larger gulls also prey on terns and small waders. Tern populations have been stable in our area. In other areas, the Mew Gull and Common Tern have increased, while the Lesser Black-backed Gull has decreased (Hario & Rintala 2008, 2011, 2014). The population of Common Redshank decreased, both on the risk and reference islets during the study years. Only the Common Ringed Plover population increased slightly, especially on the restricted-reference islets. The Archipelago Bird Census revealed that the Common Ringed Plover has moderately increased and the Ruddy Turnstone and Common Redshank have moderately decreased in the coastal areas of Finland during 1986--2013 (Hario & Rintala 2014).

Nesting archipelago birds are more often disturbed on open-access islets than in restricted-access conservation or military areas (Matti Luostarinen, personal observations). The population of Common Ringed Plover was increasing on restricted access islets in the reference area, where disturbance is lowest. Also the density of European Herring Gull population was highest there. The trends of Ruddy Turnstone and Great Black-backed Gull populations were somewhat more negative in the risk than in the reference areas. This may have been a random result, or these species may be more sensitive to human disturbance than others. Most species showed no difference between trends on the risk and reference islets. Other studies have shown that in Finland more Common Eiders breed in protected areas than would otherwise be expected considering the total extent of these areas (Kilpi 1997). Protected archipelago areas will be even more crucial to conservation of future seabird populations.

Our analysis revealed no clear evidence that the populations of the species studied have undergone severe stress from the environmental changes resulting from the Vuosaari Harbour project. Most of the population changes were probably associated with processes acting over larger areas, which thus affected the overall populations and may also have masked processes arising from local disturbances.

However, the type of islet also affected the temporal and spatial patterns of certain species. Both area and islet type affected some populations, particularly the patterns of highly colonial species, such as gulls and terns. The fact that our study did not reveal negative impacts caused by construction of the harbour to most of the monitored species does not necessary rule out disturbance effects. Examining only the number of pairs does not necessarily enable estimation of direct individual impacts. When populations increase, poor-quality areas may also be occupied, often by young and unexperienced individuals. This can prevent determination of the real effects of construction. The poor breeding success may have been masked by individuals that originated from other areas. In the future we need to develop better monitoring methods, to monitor population changes, but also individual reproduction and survival rates of nearby breeding birds during large construction projects.

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Sammanfattning

I början av 1990-talet inleddes planeringen av den nya Nordsjö Hamn i Helsingfors och 1996 fastställdes planen, vilket innebar att hamnen kom att förläggas nära ett Narura 2000-område. Det fanns farhågor om att arbetena skulle innebära störningar av fågellivet såväl i hamnområdet som längs den farled som skulle konstrueras från hamnen till öppet hav. Farleden passerade ett stort antal öar och skär med häckande fåglar. För att undersöka om de förmodade störningarna hade inverkan på fågellivet genomfördes åren 2001–2011 en övervakning av sjöfåglarna. Undersökningen omfattade 38 fågelskär. Eftersom man förmodade att risken för störningarna inte borde sträcka sig längre ut från hamn och farled än två kilometer uppdelades skären i två kategorier (Figur 1). De innanför två kilometer betraktades som skär med störningsrisk och de jämfördes med de skär som låg utanför som referens. Skären inventerades normalt tre till fyra gånger per år. Tidpunkterna var anpassade till olika arter och faser i häckningsförloppet. Besöken var korta, 15–30 minuter, eftersom längre besök skulle ha kunnat skada fågellivet. Antal par som registrerades framgår av Appendix 1, separat för riskskären och referensskären.

Vilka faktorer som påverkade fågellivet analyserades statistiskt med hjälp av modellering, och de signifikanta faktorerna redovisas i Tabell 1. I Figur 2 jämförs de modellerade beståndsförändringarna mellan risk- och referensområdena.

Totalt registrerades 28 häckande arter med mellan 980 och 1196 par på skären i riskområdet och 32 arter med mellan 1090 och 1615 par i referensområdet. Om enskilda arter kan följande nämnas. Roskarl och havstrut hade mer negativa trender i riskområdet än i referensområdet. Knölsvan minskade generellt och hade lägre täthet i referensområdet. Kanadagås ökade i referensområdet men inte i riskområdet. Vitkindad gås ökade något snabbare i referens- än i riskområdet. Gräsand minskade i båda områdena men hade signifikant högre täthet i referensområdet. Vigg minskade signifikant men särskilt mycket i referensområdet. Ejder uppvisade exponentiell tillväxt i båda områdena. Även strandskata ökade i båda områdena. Rödbena minskade

något, särskilt i riskområdet. Större strandpiparens täthet var större i referensområdet än i riskområdet. Inventeringarna visade på störst förändringar för kolonilevande arter såsom måsar. Gråtruten minskade men hade högst täthet på skär med besöksrestriktioner. Silltruten ökade på risköarna medan fiskmåsen ökade på referensöarna. Tärnorna (arterna sammanslagna) varierade mycket i antal men variationen hade inget samband med omvärldsfaktorerna.

Flera av förändringarna i undersökningsområdet är desamma som längs Finlands kust i övrigt, nämligen för kanadagås, vitkindad gås, vigg, rödbena, roskarl och gråtrut. Både havstrut och gråtrut har minskat generellt, troligen som en effekt av soptipparnas försvinnande. Dessa arters predation kan vara förklaringen till att de mindre måsarna, fiskmås, silltrut och skrattmås, kunnat öka i antal. De större måsarna konsumerar också ägg och ungar av tärnor, men tärnornas antal har inte reducerats i området.

Sammanfattningsvis visade vår analys på betydande variation mellan olika arters reaktion på eventuella störningar och effekter av generella storskaliga populationsprocesser. Vi kunde inte påvisa några entydiga eller systematiska effekter av hamn- och farledsbyggandet. De flesta beståndsförändringarna var troligen resultat av processer som verkade över större områden. Vi kan dock inte utesluta att dessa processer kan ha dolt vissa lokala störningseffekter.

Appendix 1. Yearly number of pairs. *Årligt antal par.*

Risk Islets						Year					
Species	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Cygnus olor	7	11	4	5	7	6	6	9	7	5	8
Branta canadensis	3	3	2	3	2	3	7	4	2	3	3
Branta leucopsis	2	4	6	11	11	19	24	29	46	35	41
Anas penelope	1	0	0	0	0	0	0	0	0	0	0
Anas platyrhynchos	5	5	4	5	5	3	2	2	2	2	1
Anas clypeata	0	2	2	1	2	0	1	0	2	2	0
Aythya fuligula	23	24	18	19	13	24	4	14	21	17	19
Somateria mollissima	114	109	103	118	121	92	109	126	101	139	161
Melanitta fusca	0	0	0	0	0	0	0	0	0	0	1
Bucephala glangula	0	0	0	0	0	0	0	0	0	0	0
Mergus serrator	1	0	0	0	0	1	0	0	0	1	0
Mergus merganser	0	2	1	0	0	0	0	0	0	0	0
Podiceps cristatus	0	0	0	0	0	0	0	0	0	0	0
Haematopus ostralegus	6	4	4	6	6	7	6	9	8	7	6
Charadrius hiaticula	2	1	2	1	1	1	2	3	1	2	5
Actitis hypoleucos	0	0	0	0	1	1	1	0	2	2	1
Tringa totanus	5	5	3	3	1	0	3	4	2	2	2
Arenaria interpres	5	5	4	3	4	1	4	1	3	3	1
Larus ridibundus	280	277	385	386	285	184	163	87	95	156	78
Larus canus	263	171	278	285	349	286	288	346	359	345	382
Larus fuscus	0	0	0	0	1	1	1	2	2	4	4
Larus argentatus	151	196	196	195	180	147	147	116	138	118	102
Larus marinus	8	9	8	7	6	5	2	4	3	3	3
Hydroprogne caspia	2	1	1	2	1	1	2	1	1	1	0
Sterna hirundo	1	1	0	22	0	1	1	0	0	1	2
Sterna paradisaea	20	17	18	6	12	8	16	15	20	15	76
Sterna hirundo/paradisaea	180	136	140	85	91	166	181	200	191	130	135
Cephus grylle	0	0	0	0	0	0	0	0	0	0	0
Anthus petrosus	3	2	4	1	3	3	4	3	3	4	2
Motacilla alba	11	8	7	13	14	12	11	9	12	11	10
Oenanthe oenanthe	8	3	5	0	8	6	8	6	5	5	6
Corvus corone cornix	0	0	1	2	2	2	2	1	1	1	1
Total	1101	996	1196	1179	1126	980	995	991	1027	1014	1050

Appendix 1. Yearly number of pairs. *Årligt antal par.*

Reference Islets	Year										
Species	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Cygnus olor	9	9	1	6	7	9	8	12	10	9	6
Branta canadensis	3	2	5	3	3	4	6	6	5	6	5
Branta leucopsis	0	4	13	24	28	45	54	59	80	72	91
Anas penelope	1	0	1	0	1	1	0	1	1	1	2
Anas platyrhynchos	10	8	10	6	5	5	1	4	2	1	0
Anas clypeata	1	0	1	0	1	0	0	0	1	1	2
Aythya fuligula	54	32	55	40	24	42	23	29	40	40	23
Somateria mollissima	162	134	171	153	139	139	139	150	156	204	265
Melanitta fusca	0	0	0	1	0	0	0	0	0	1	1
Bucephala glangula	1	0	1	2	1	1	1	1	0	1	0
Mergus serrator	0	1	0	0	1	1	1	0	2	2	1
Mergus merganser	0	0	1	2	1	2	2	1	1	1	3
Podiceps cristatus	0	0	0	0	0	0	0	0	1	0	0
Haematopus ostralegus	6	8	5	8	8	4	3	9	11	9	8
Charadrius hiaticula	4	3	1	2	1	0	1	4	2	2	1
Actitis hypoleucos	0	0	1	0	2	0	0	0	1	0	0
Tringa totanus	6	4	6	6	5	6	5	5	7	4	5
Arenaria interpres	9	3	5	4	5	2	5	6	6	5	3
Larus ridibundus	0	0	0	1	0	0	0	21	58	120	300
Larus canus	347	328	390	384	411	349	419	411	404	408	426
Larus fuscus	1	2	3	3	3	3	3	2	1	2	2
Larus argentatus	189	199	239	251	249	209	200	184	146	124	102
Larus marinus	10	8	8	9	7	8	10	6	6	8	6
Hydroprogne caspia	3	2	2	3	2	1	1	1	1	1	2
Sterna hirundo	22	30	20	44	20	33	15	20	2	4	20
Sterna paradisaea	8	6	0	0	1	17	0	5	2	1	0
Sterna hirundo/paradisaea	390	250	251	168	288	388	436	475	529	364	275
Cephus grylle	24	22	16	26	17	20	26	35	37	24	23
Anthus petrosus	8	6	7	6	9	7	10	9	8	8	9
Motacilla alba	19	18	14	17	20	15	20	22	21	17	22
Oenanthe oenanthe	11	11	5	1	11	5	11	7	8	7	12
Corvus corone cornix	1	0	0	0	1	0	0	0	0	1	0
Total	1299	1090	1232	1170	1271	1316	1400	1485	1550	1450	1615

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Changes in numbers and distribution of wintering Long-tailed Ducks *Clangula hyemalis* in Swedish waters during the last fifty years

Förändringar i antal och utbredning hos övervintrande alfåglar Clangula hyemalis *i svenska farvatten under de senaste femtio åren*

LEIF NILSSON

Abstract

The Long-tailed Duck *Clangula hyemalis* is the most common wintering waterfowl in Swedish marine waters but it is not well covered in the International Midwinter Counts due to its offshore distribution making special surveys necessary for the species. Surveys covering the entire Baltic Sea were organized for the first time in 1992/1993 with follow-up surveys in 2007–2011 and 2016. In Sweden, aerial line transect surveys covered all areas of importance for the Long-tailed Duck during the last two all-Baltic surveys. In the 1970s extensive regional surveys were done from coastguard boats and airplane. The total wintering population of Long-tailed Duck was estimated to be around 370,000 individuals in 2016 compared to 436,000 in 2009, and in the order of 1,400,000 in 1992/1993, indicating an ongoing decrease. Comparing the more recent surveys with the surveys from the 1970s, the most marked changes in wintering numbers were found east of Gotland, where only a small proportion of the numbers estimated in the 1970s remained. Decreases were also found in the waters around Öland, whereas no marked changes were noted for the smaller wintering populations of the east coast archipelagos.

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Introduction

The Long-tailed Duck *Clangula hyemalis* is the most common wintering waterbird in Swedish waters but due to its marine distribution it is not so well covered in the Midwinter Counts organized annually since 1967 (Nilsson 1972, 1975, 1980, 2012, Nilsson & Haas 2016). Proper monitoring of the Long-tailed Duck requires special surveys in the archipelagos and the offshore areas.

During the years 1965–1978 more than 100 boat surveys were undertaken along the Swedish coasts from the patrol boats of the Swedish Coast Guard (Nilsson 1972, 1980). Extensive aerial surveys were also undertaken mainly in the archipelagos but also along the open coasts of Scania, Öland and Gotland (Nilsson 1975, 1980). It was however not possible to survey the Midsjö Banks and Hoburg Bank far out in the middle of the Baltic Sea even if these areas were known to be sites for large numbers of wintering Long-tailed Ducks.

In the early 1990s, after the political changes in the region, it was for the first time possible to organize extensive offshore surveys of the wintering waterbirds in the entire Baltic in 1992–1993. Maps and population estimates were produced for all important waterbirds (Durinck et al. 1994). A second survey of the entire Baltic was organized in 2007–2010, the SOWBAS-project (“Status of Wintering Waterbird Populations in the Baltic Sea”; Skov et al. 2011).

Analysis of the results from the SOWBAS-project showed a decrease in the wintering population of Long-tailed Duck in the Baltic of about 65% since the surveys in 1992/1993, with important decreases also for other wintering seaducks (Skov et al. 2011). Marked decreases in numbers of the species were documented from all parts of the Baltic, including Sweden (Nilsson 2012). As a response to this decrease the species was red-listed and work with a single-species action plan was started by AEWA (African European Waterfowl Agreement).

To follow up the situation for the wintering waterbirds in the area a third all-Baltic survey was organized in the winter 2016. In the last few years the Midwinter Counts have been used as an important part of the indicator “Numbers and trends in breeding and non-breeding marine water birds” which is now accepted as a Core Indicator among a set of other indicators to monitor the situation in the marine areas of the European Union as a part of the MSFD (Marine Strategy Framework Directive). In this work offshore surveys of wintering water birds will be an important part in the future.

In this paper I analyze the results from the offshore surveys of Swedish waters in 2016 with the main focus on the Long-tailed Duck. Earlier, results from the boat surveys have been summarized by Nilsson (1980), whereas the aerial surveys under SOWBAS were presented by Nilsson (2012). The present paper updates the previous presentations of the development of the wintering populations and compares the winter distribution of the species from the last survey with the results from the previous surveys.

Material and methods

In 2016, as in the 2007–2011 surveys, the offshore areas were covered by line transects from an aircraft. The census lines were laid out so that all important water areas (including the offshore banks) in the Swedish part of the Baltic were covered out to a depth of about 30 m. The survey lines started at the shore or, in the extensive east coast archipelagos (mainly the Stockholm archipelago), from the inner larger islands. Generally, there was a distance of 4 km between the survey lines but in some special areas (e.g. Hanö Bight and Falsterbo) there was a distance of 2 km between the lines. The survey lines are shown in the distribution maps from the surveys (Figure 2 for an overview and the regional maps).

For the surveys a Cessna 337 Skymaster, a twin-engined high-winged aircraft with good visibility was used. The flying altitude was about 70 m and the speed 150–180 km/h, i.e. the slowest possible. Aerial surveys were only undertaken in good weather conditions. Fixed waypoints at the ends of each transect were established and navigation was undertaken with the aid of the GPS of the aircraft. Another GPS recorded the actual flight track taking positions every ten seconds. Two observers covered each side of the aircraft. All observations were recorded on tape with time and were later transferred to a database with the positions from the GPS recordings.

During the surveys in 2007–2011, all water birds were counted within a survey belt extending 200 m on either side of the aircraft. This means that we covered a zone 320 m wide as there was a dead angle below the aircraft. During SOWBAS distance sampling was not used but observations outside the survey belt was recorded as additional information. During the 2016 survey the data were recorded in three different bands to make the data collection more similar to the methods used in the other countries: A (=main band) out to 200 m, B 200–500 m and C 500–1000 m. In 2007 – 2011, counts were only made in the main band.

The counts from the line transects were used to estimate regional totals for the different species using the counts within the main band of the surveys and a factor based on the coverage of the different regions (6.25 and 12.5, respectively for transects separated by 2 and 4 km). For some areas with very marked differences in densities of different species between areas a stratified sampling routine separating low and high density areas was used in the calculations. Even if data were collected from three different bands in 2016, I only used the data from the main band in the calculations presented here to make the results from the surveys in 2007–2011 and 2016 comparable with each other.

In 2016, the offshore areas from Falsterbo in the southwest to the Gävle Bight in the north were covered with transect lines (Figure 2). In the south, some transects in the Öresund could not be surveyed due to flying restrictions in the Control Area of Copenhagen airport. Moreover a few lines in the Stockholm archipelago were not flown as planned partly due to military activities. The differences in coverage was compensated for in the calculations.

The total area was surveyed during ten days between mid-January and mid-March 2016. The total flying time was 65 hours, of which 43 hours were used for actual bird counts. The 2016 winter was mild and there was no ice in the outer marine areas during the survey and only some ice in the inner parts of the archipelagos.

The surveys in the archipelagos in 2016 only covered the outer part with potential areas for species like the Long-tailed Duck. The inner parts of the archipelagos were covered by aerial surveys in the winter 2015. The surveys in the Stockholm archipelago 2015 covered the entire survey lines from the mainland coast to the outer skerries and will be included in the analysis presented here when it comes to distribution. For information about the earlier surveys used in the comparisons see Nilsson (1980, 2012).

Results

The national picture

At the first total survey of the Baltic in 1992/1993 (Durinck et al. 1994), the wintering population of Long-tailed Duck in Swedish waters was esti-

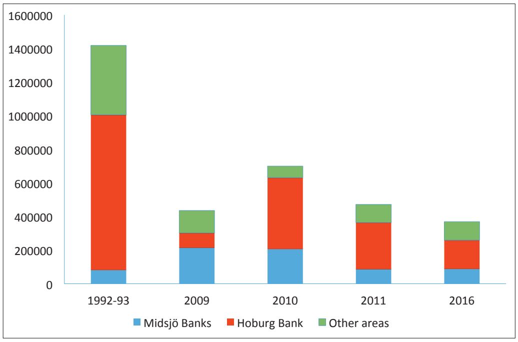


Figure 1. Estimated total numbers of wintering Long-tailed Ducks *Clangula hyemalis* in Swedish waters at the country-wide surveys shown separately for the important offshore banks and the other areas. A regional break-down is presented in Table 1.
Beräknat antal övervintrande alfåglar Clangula hyemalis I svenska vatten vid de landstäckande inventeringarna uppdelat på de viktiga bankarna samt övriga områden. En uppdelning på olika regioner återfinns i Tabell 1.

mated to be 1.4 million (Figure 1, Table 1). At the SOWBAS survey in 2009, the estimated total for Sweden was 436,000 individuals. As for the entire Baltic (Skov et al. 2011) this was a very marked reduction of the population. Numbers were higher at the 2010 survey with an estimate of 700,000 individuals. 2010 and 2011 were however hard ice winters and large important areas in the eastern part of the Baltic like the Riga Bay and Irbe Straits were ice-covered and the Long-tailed Ducks forced to move.

At the survey in 2016, the estimated population was lower still with 370,000. Lacking data yet from the other countries it is difficult to establish whether there was a real decrease in wintering numbers or just a shift in the distribution to other areas in the Baltic.

In the mild winters of 2009 and 2016, the Long-tailed Ducks were found all along the surveyed part of the Swedish Baltic coast (Figure 2). There was however a marked concentration to the three large offshore banks, Södra Midsjöbanken, Norra Midsjöbanken and Hoburgs Bank, where about 70% of all Long-tailed Ducks were found in both years. On the other hand, the Long-tailed Ducks were much

Table 1. Estimated totals of wintering Long-tailed Ducks *Clangula hyemalis* in different areas along the coasts of Sweden.
Uppskattat antal övervintrande alfåglar Clangula hyemalis *in different areas along the coasts of Sweden.*

Area	1970-74	1992-93	2007	2009	2010	2011	2016
Falsterbo + S Öresund		500		1600	2800	1200	900
Scania south coast	10000	800		1700		2000	4300
Scania southeast		200		100		300	400
Hanö Bight	25000	17000	23000	17000		7100	7200
Blekinge archipelago	1600	1100	300	100		100	200
Kalmarsund		12000	23000	11000	11500	2500	5800
Öland east coast	40000	10000	19000	26000		39000	9100
Ölands northern banks	60000	30000	11000	5000	7200	22400	3500
Midsjö banks		81000		213000	206000	85000	87200
Hoburgs bank		925000		90000	426000	280000	173000
Gotland east coast	400000	270000		11000	15100	15700	11800
Gotska sandön + northern banks	20000	10000			13500	14500	3900
Gotlands west coast)		23000		2000		2000	6700
Kalmar archipelago (N Kalmarsund)	10000	12000		14000	2700	Ice	1600
Österg archipelago	1000	3500		8800	3200	Ice	6100
Sörml. archiepalo	4000	4000		4100	12000	Ice	2400
Stockholm archipelago	24000	18000		26100		Ice	37000
Uppland Northern coast				3700		Ice	2600
Gävle Bight				600		Ice	5800
Total		1418100		435800	700000	471800	369500

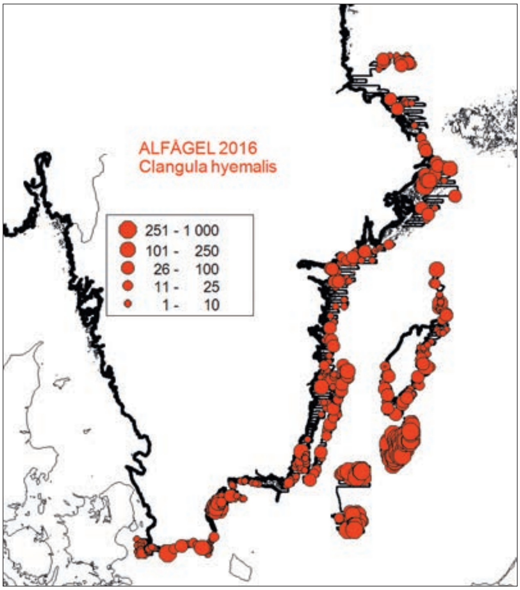


Figure 2. The distribution of Long-tailed Ducks *Clangula hyemalis* in Swedish waters during the winter 2016. Black lines in this and following maps show the flight tracks of the aerial surveys.
Utbredning för alfågel Clangula hyemalis i svenska farvatten vintern 2016. Svarta linjer i denna och följande kartor visar den aktuella rutt som flugits.

more concentrated in the ice-winters 2010 and 2011 (Nilsson 2012), with 90% seen on the banks in 2010, a winter with heavy ice coverage in the Baltic.

Regional pattern

Small groups and a few larger flocks were found in the southern parts of the Öresund and along the south and southeast coasts of Scania (Figure 3). The total estimate for the area was 4300 in 2016 to be compared to about 10,000 or more during the 1970s. Three surveys during the SOWBAS-survey showed similar distributions (Nilsson 2012). During the sixties and seventies the species was more common in the area with up to a few thousand also in inshore waters (Nilsson 1972, 1980).

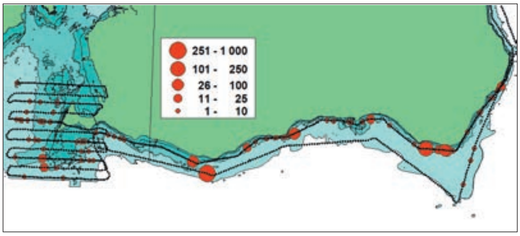


Figure 3. Distribution of Long-tailed Ducks *Clangula hyemalis* at the south coast of Scania in 2016.
Utbredning för alfågel Clangula hyemalis vid den skånska sydkusten 2016.

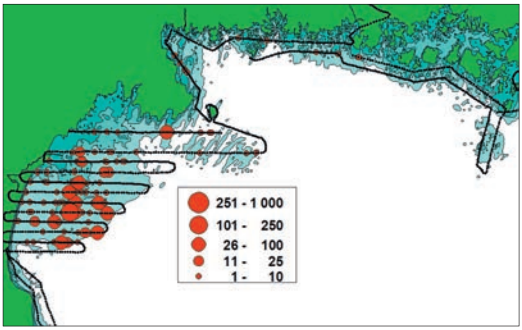


Figure 4. Distribution of Long-tailed Ducks *Clangula hyemalis* in the Hanö Bight in 2016.
Utbredning för alfågel Clangula hyemalis i Hanöbukten 2016.

The Hanö Bight has been well-known as the southernmost concentration area for Long-tailed Ducks in Sweden with the majority of the birds concentrated to vast shallow areas in the western offshore part of the bight (Figure 4), whereas only very few individuals were seen in the outer parts of the Blekinge archipelago further to the east. During the early years of the study about 25,000 Long-tailed Ducks were estimated to winter in the Hanöbukten area but numbers decreased to about 7000 during the last two surveys. During the sixties and seventies flocks were also regularly seen in the outer parts of the Blekinge archipelago with an estimated total of about 2000 but in more recent years, numbers were appreciably lower here (Table 1).

The Kalmarsund and especially the waters east and north of Öland are important areas for wintering Long-tailed Ducks (Figure 5). Numbers in the Kalmarsund have been quite variable between 2500 and 23,000 during the recent surveys but the area will quite easily get ice-covered (Table 1). The areas east and north of Öland belong to the more important winter areas for the species close to the mainland. Rough estimates from the 1970s indicate that the population then could have been as high as 100,000, but recent estimates are much lower and in 2007–2016 it varied between 13,000 in the mild winter of 2016 and 61,000 during the cold ice-winter 2011, when Kalmarsund and most archipelagos were ice-covered.

As stated above the offshore banks, Midsjö Banks and Hoburg Bank, are the most important Long-tailed Duck areas in Swedish waters with

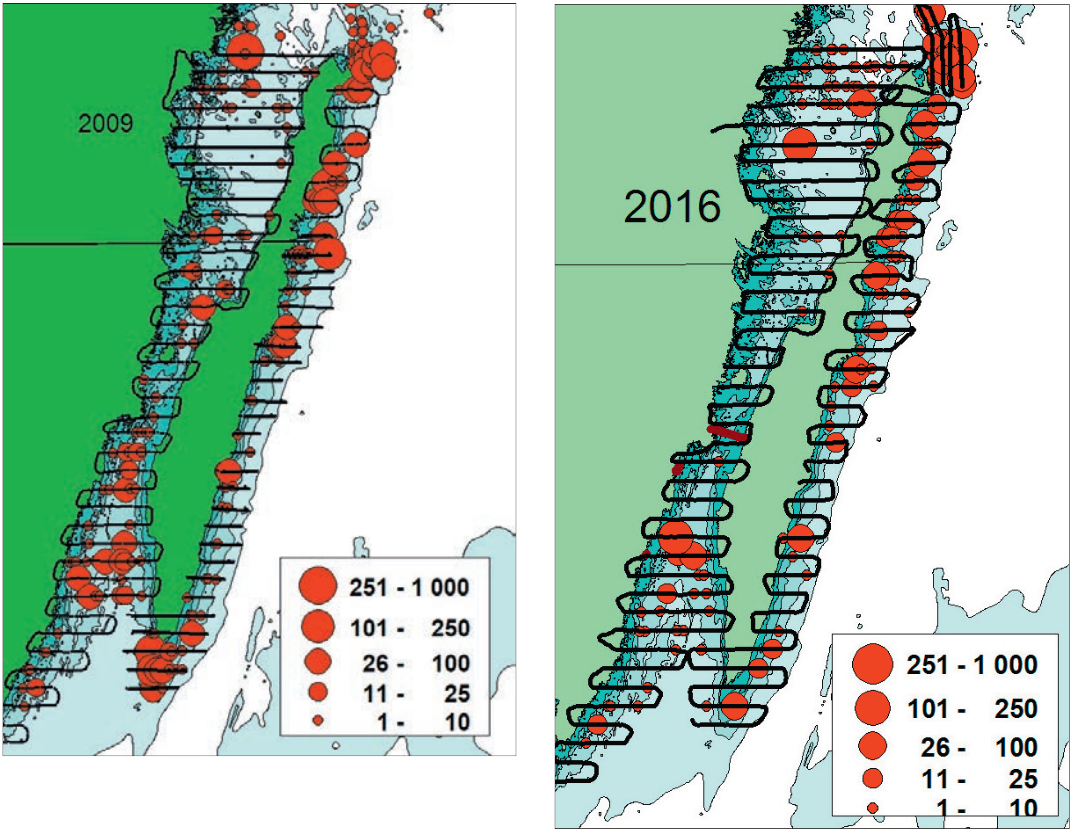


Figure 5. Distribution of Long-tailed Ducks *Clangula hyemalis* in the Kalmarsund and around Öland in the winters of 2009 and 2016.
Utbredning för alfågel Clangula hyemalis i Kalmarsund och runt Öland vintrarna 2009 och 2016.

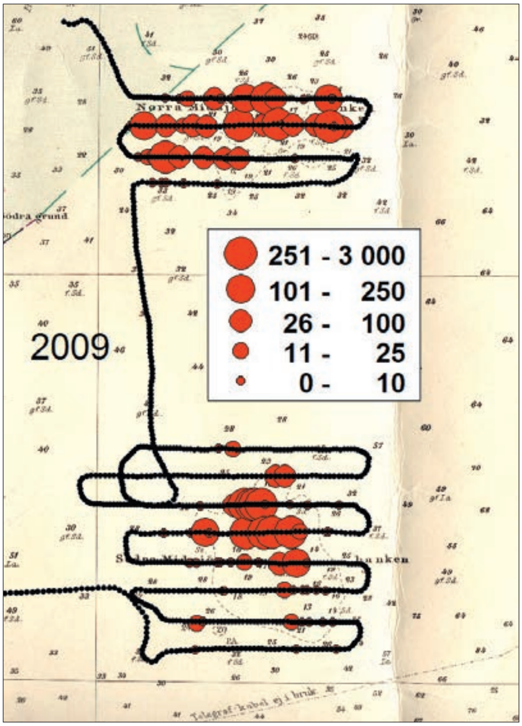
about 70% of the total Swedish wintering population in normal winters and up to 90% during one of ice winters. The local distribution of the Long-tailed Ducks on the banks are illustrated by the counts in the normal winters (Figure 6 for the Midsjö Banks and Figure 7 for Hoburg Bank).

During the years 2009–2016, the Midsjö Banks have been covered by aerial surveys at six different occasions with the total numbers varying between 44,500 and 213,000 individuals (Table 2). At the first Baltic survey in 1992/1993 the number of Long-tailed Ducks on the Midsjö Banks was estimated to be 81,000. For Hoburg Bank the estimates for 2009 to 2016 varied between 90,000 and 426,000, the highest total was obtained for the ice-winter 2010, when the east coast of the Baltic was mostly ice-covered. In 1992/1993 the estimate for Hoburg Bank was no less than 900,000 Long-tailed Ducks. Unfortunately, there are no older censuses available from these important off-shore banks.

The coasts of Gotland are important areas for wintering Long-tailed Ducks (Figure 8) with many flocks occurring along all parts of the island except at the steep coasts of NW Gotland where flocks are fewer and smaller. The banks between northern Gotland and Gotska Sandön have historically been important for the Long-tailed Ducks.

During the years 2009 to 2016 the estimates from four surveys has varied between 11,000 and 16,000 for the entire east coast. The banks north of Gotland were surveyed in 2010, 2011 and 2016 with 13,500 and 14,500 Long-tailed Ducks found for the first two winters but less than 4000 in 2016. At the surveys in 1992/93 no less than 270,000 were estimated for the east coast of Gotland.

Unfortunately, there are very few historical data from the offshore areas east of Gotland but there is some information from a few coast guard patrols in the area. At one of these patrols to the areas around Fårön in NE and neighboring areas, the observers



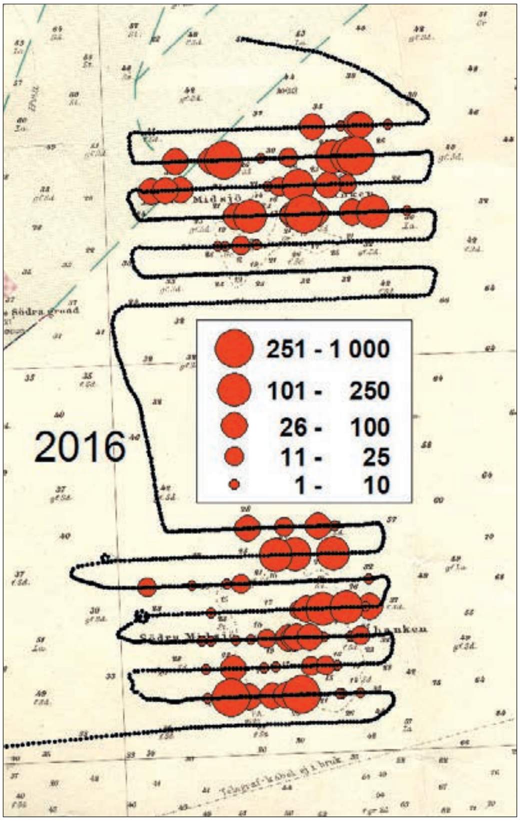


Figure 6. The distribution of Long-tailed Ducks *Clangula hyemalis* on the Midsjöbanks during the mild winters 2009 and 2016.
Utbredning för alfågel Clangula hyemalis på Midsjöbankarna under de milda vintrarna 2009 och 2016.

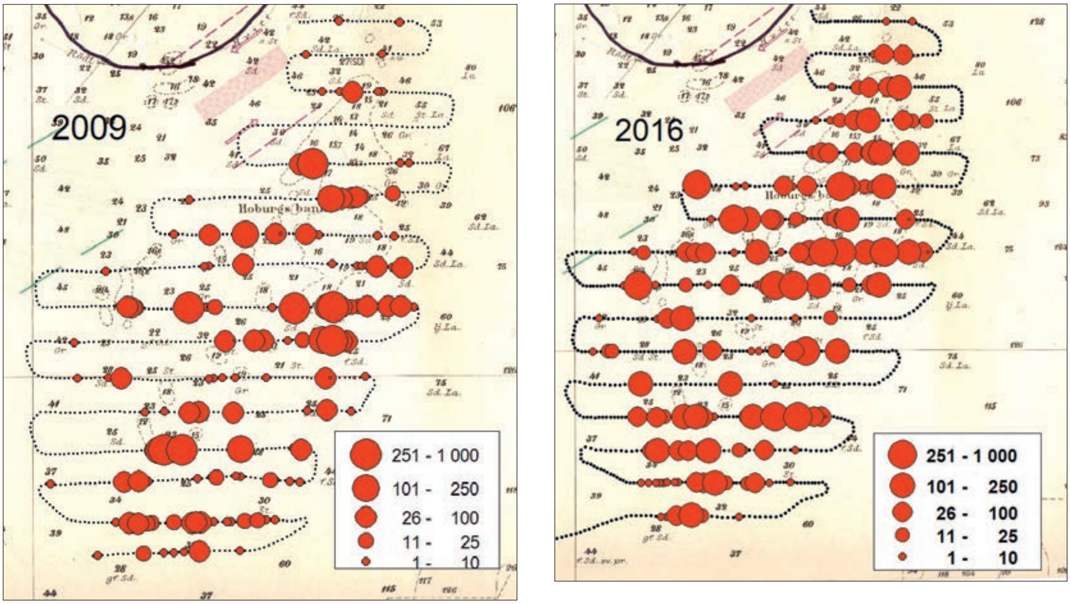


Figure 7 The distribution of Long-tailed Ducks *Clangula hyemalis* on Hoburgs Bank during the mild winters 2009 and 2016
Utbredning för alfågel Clangula hyemalis på Hoburgs Bank under de milda vintrarna 2009 och 2016.

Table 2. Estimated totals of wintering Long-tailed Ducks *Clangula hyemalis* on the two Midsjö Banks at the different occasions when the banks were surveyed from the air.
Uppskattat antal övervintrande alfåglar Clangula hyemalis *vid Midsjöbankarna vid de olika tillfällen bankarna flyginventerats.*

	2005-04-01	2009-03-03	2010-03-16	2011-03-07	2011-03-29	2011-04-20	2016-02-24
Northern		76000	74000	63000	37000	67000	56700
Southern	160000	137000	132000	22000	7500	16500	34500
Total		213000	206000	85000	44500	83500	91200

actually counted 70,000 from the ship. Based on densities obtained from the coast guard patrols and one aerial reconnaissance flight in 1974, the wintering population of the Long-tailed Duck in the offshore areas east of Gotland is roughly estimated to have been in the order of 400,000 in the early seventies (Nilsson 1980). The total numbers wintering here in recent years may be as low as 5% of the numbers during the early seventies.

The Long-tailed Duck is widely distributed in the archipelagos from Kalmarsund in the south up to and including the Stockholm archipelago (Figures 9–12). These areas were extensively covered from Coast Guard Patrols and area-based aerial surveys in the 1970s as well as by the aerial transects in later years. This offers good possibilities to compare the situation in the two time periods 1971–1978 and 2009–2016.

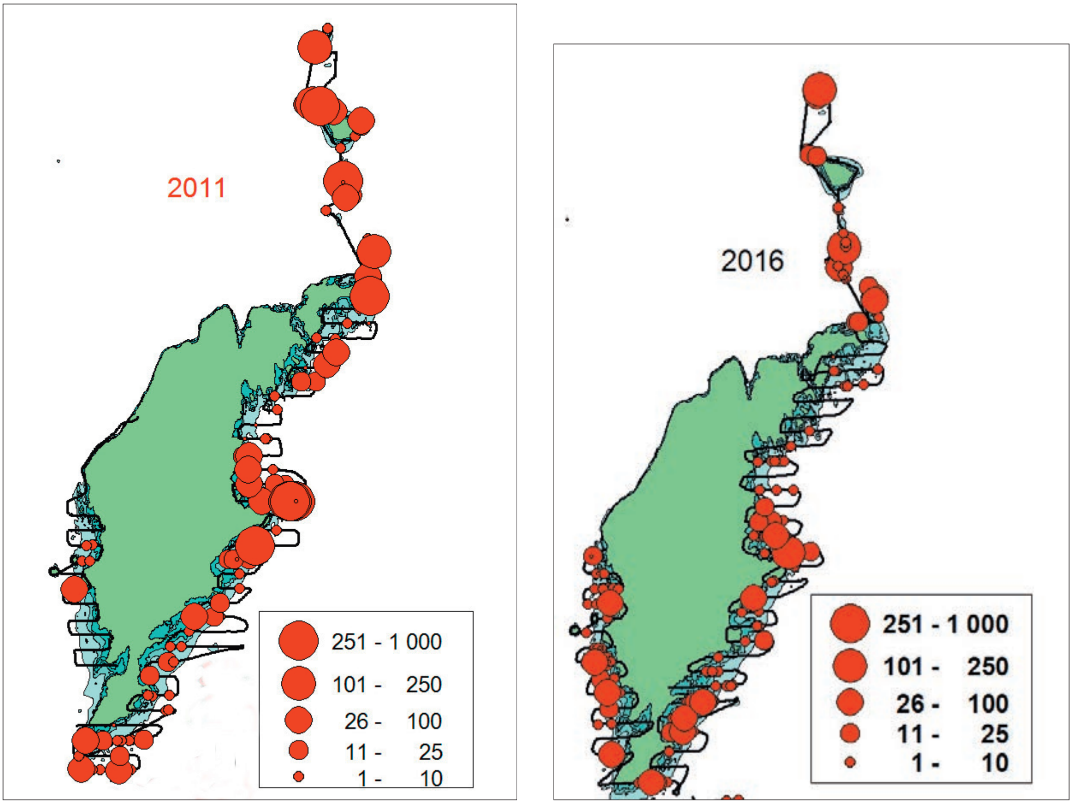


Figure 8. The distribution of Long-tailed Ducks *Clangula hyemalis* around Gotland in the winters 2011 and 2016.
Utbredning för alfågel Clangula hyemalis runt Gotland vintrarna 2011 och 2016.

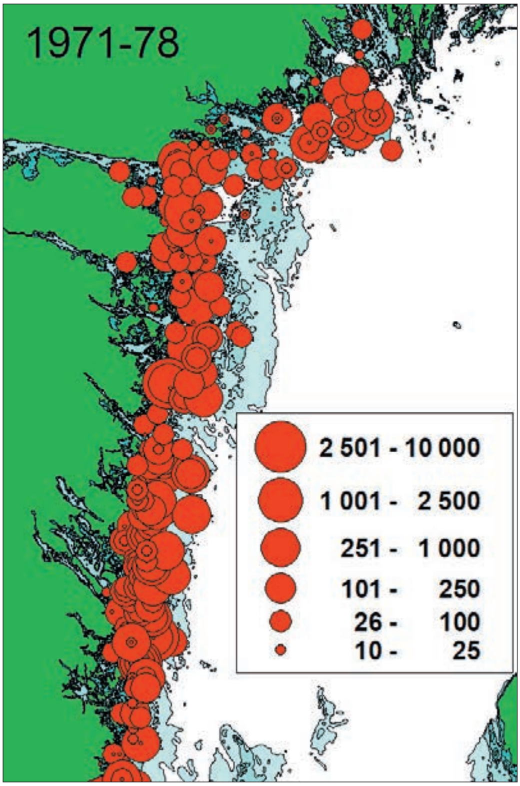


Figure 9. Distribution of wintering Long-tailed Ducks *Clangula hyemalis* in the archipelago areas of N. Kalmar, Östergötland and Södermanland counties. The map is based on accumulated boat and aerial surveys during 1971 – 78. *Utbredning för övervintrande alfåglar Clangula hyemalis i skärgårdarna av norra Kalmar, Östergötland och Södermanlands län. Kartan baseras på ackumulerade båt och flyginventeringar 1971 – 78.*

The archipelagos of Kalmar, Östergötland and Södermanland counties (Figures 9, 10) are rather narrow. In the 1970s Long-tailed Ducks were commonly found in the middle and outer parts of the archipelagos and the accumulated map (Figure 9) show occurrences of flocks along the entire coastline. In the later surveys in 2009 and 2016, flocks were found in all parts in 2009, whereas hardly any Long-tailed Ducks were found in some parts of the area in 2016. Compared to the 1970s, the Long-tailed Ducks were more common in the outer parts of the island zone with fewer in the zone with skerries further out.

There were no marked changes in numbers between the seventies (15,000), 1992/1993 (19,000) and 2010 (18,000) for these parts of the archipelagos (Table 1). On the other hand, about 27,000 were estimated for the area based on the aerial surveys in 2009 compared to less than 10,000 in 2016.

In contrast with the archipelagos discussed in the previous paragraphs, the Stockholm archipelago is much wider with a distance of up to 70 km from the mainland coast to the outer skerries in some parts. The Stockholm archipelago was also intensively surveyed from Coast Guard patrols and area-based aerial counts during the 1970s. In later years, one area-based survey (2004) and three transect counts were made of the Long-tailed Duck areas.

Comparing the distribution patterns in the 1970s and 2004–2016 based on the accumulated totals from all counts, some marked differences were found between the two time-periods (Figure 11). In 2004–2016 much smaller numbers of Long-tailed Ducks were found in the southern parts of the archipelago compared to the 1970s. A similar difference was found for the outermost areas in the central part of the archipelago, where relatively few flocks were found in the later period. In both periods, the northern parts of the archipelago were important for the species, but more Long-tailed Ducks were found closer to the mainland in the latter period compared to the 1970s. Comparing the three years with transect surveys (Figure 12), the Long-tailed Ducks were more spread in 2009 than in 2015 and 2016, when a marked concentration was found in the central parts of the northern archipelago.

There were no clear changes in the number of wintering Long-tailed Ducks in the Stockholm archipelago between 1970–1974 and 2009, but in contrast to all other areas the highest estimate was obtained for 2016 (Table 1).

North of the Stockholm archipelago, Long-tailed Ducks were also found in winter along the coasts of Uppsala county and in the Gävle Bight (Figure 13). These areas were surveyed both in 2009 and 2016 with smaller numbers found scattered on the banks Finngrunden and in some parts of the archipelago. No historical data are available from this northernmost winter area of the species.

Discussion

The results of the censuses of Swedish offshore waters presented in this report and previous papers (Nilsson 2008, 2012) as well as the two international publications from the first two all-Baltic surveys (Durinck et al. 1994, Skov et al. 2011) clearly shows the importance of the Swedish offshore waters for the Long-tailed Duck, especially the importance of the three offshore banks: the Midsjö Banks and Hoburg Bank. The three banks together

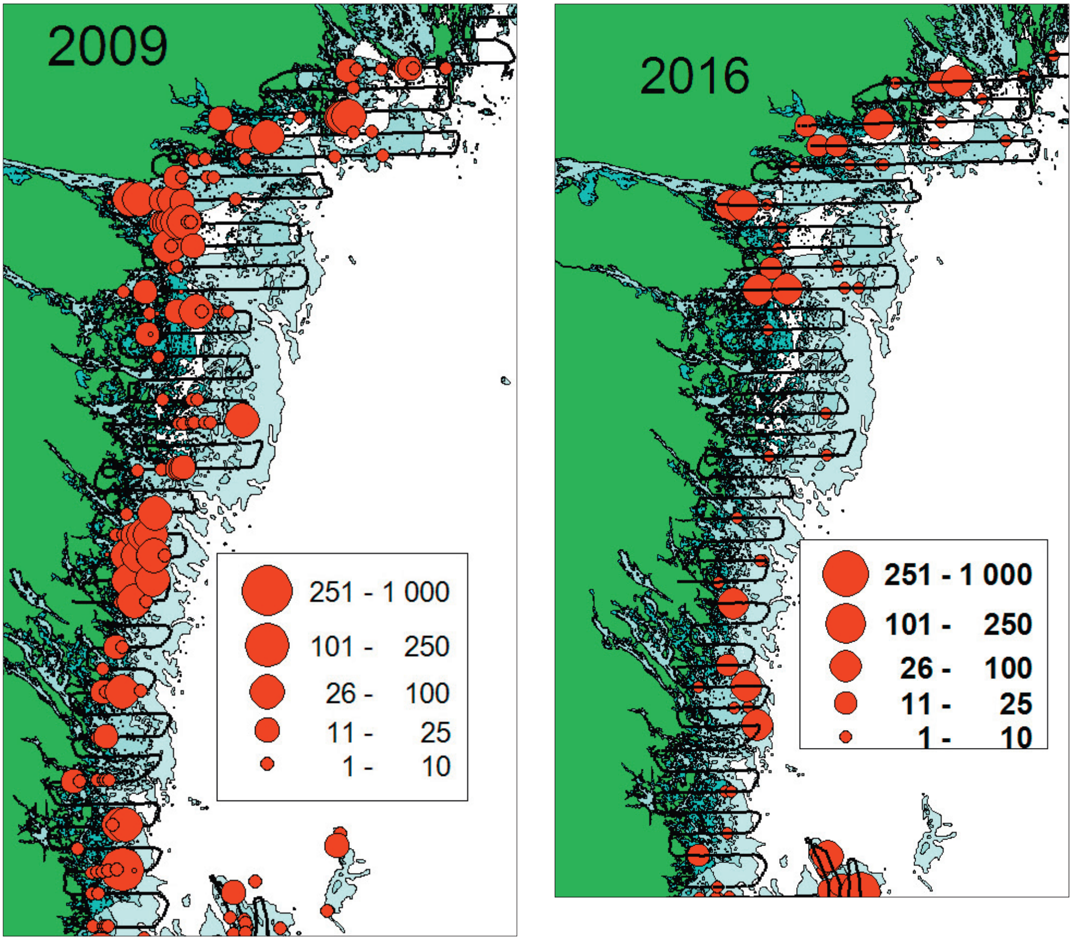


Figure 10. Distribution of wintering Long-tailed Ducks *Clangula hyemalis* based on aerial surveys in the counties of Kalmar, Östergötland and Södermanland in 2009 and 2016.
Utbredning för övervintrande alfåglar Clangula hyemalis i skärgårdarna av norra Kalmar, Östergötland och Södermanlands län. 2009 och 2016.

hold at least 70% of the Swedish Long-tailed Duck population in winter. During the first two all-Baltic surveys, the Swedish waters had about 30% of the total population, implying that the three offshore banks in the Swedish zone together harbors at least 20% of the total population and sometimes in hard winters even more (Nilsson 2012).

The previous all-Baltic survey in 2007–2009 showed a very marked decrease in the overall wintering population of Long-tailed Ducks since the first survey in 1992/1993 (Skov et al. 2011). Similarly, a marked decrease was found in the wintering populations in Swedish waters (cf. also Nilsson 2012). Between 2009 and the survey in 2016, the decrease continued and the Swedish wintering population was about 15% lower than in 2009. Until the international counts have been analyzed and made available it would however be premature to try to establish if this was a part of a general pattern or an effect of a redistribution.

In spring, the Long-tailed Ducks migrate to the breeding areas in northern Russia, a major part of the population passing the Gulf of Finland during their migration, large numbers of migrating Long-tailed Ducks being counted from sites in Finland and Estonia during a restricted period (Bergman 1974, Hario et al. 2009). Counts at Söderskär on the Finnish coast showed an increase in the number of passing Long-tailed Ducks from about 1970 to the early 1990s, followed by a marked decrease (Hario et al. 2009, see also Kauppinen & Leivo

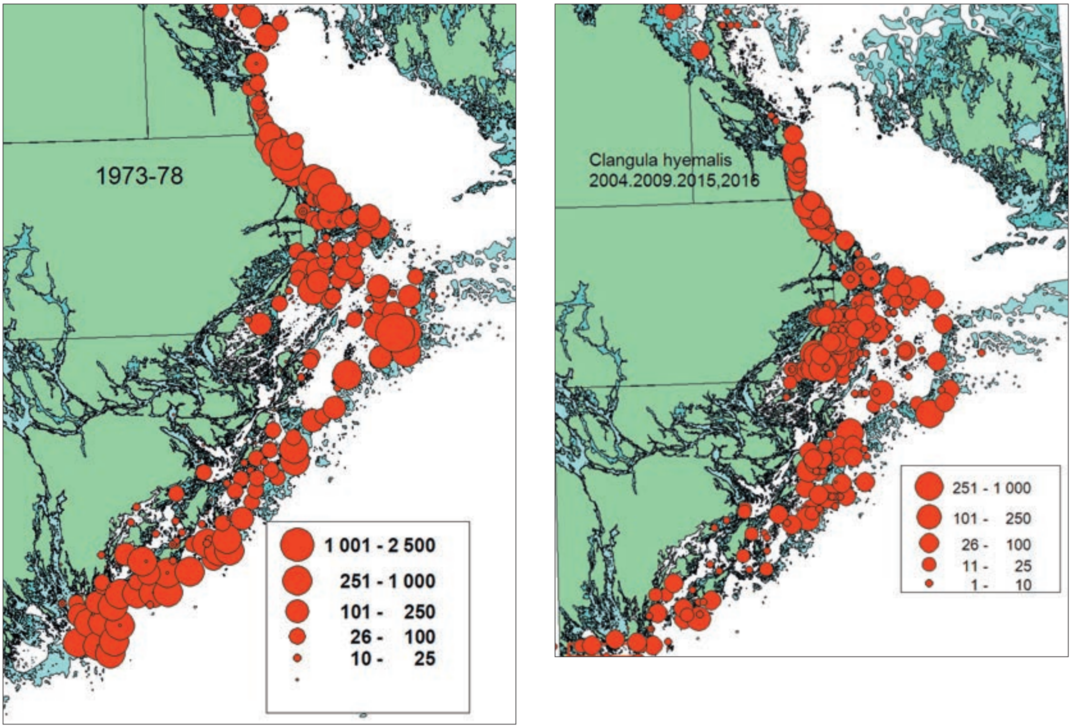


Figure 11. Distribution of wintering Long-tailed Ducks *Clangula hyemalis* in the Stockholm archipelago area. The maps are based on accumulated boat and area-based aerial surveys during 1971 – 78 and accumulated aerial surveys 2004 - 2016. Note that aerial surveys in 2004 was area-based (cf. Nilsson 2008), whereas the surveys in 2009 – 2016 were transects (details seen Figure 12).

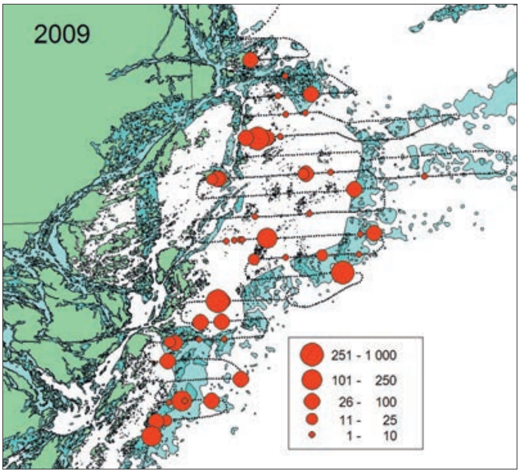
Utbredning för övervintrande alfåglar Clangula hyemalis i Stockholms skärgård. Kartorna baseras på ackumulerade båt och områdes-baserade flyginventeringar 1971 – 78och ackumulerade flyginventeringar 2004 – 2016. Observera att flyginventeringarna 2004 (jfr Nilsson 2008) var områdesbaserade, medan inventeringarna 2009 – 2016 var linjetaxeringar (detaljer se Figur 12).

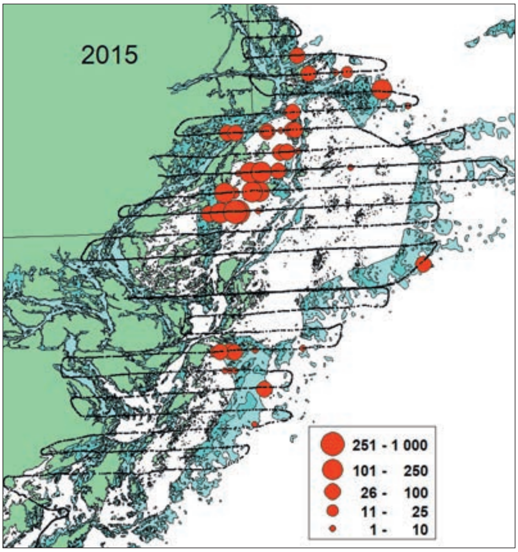
2008). The highest counts of spring migrants coincided with the first all Baltic survey (1992/1993), which yielded much higher estimates than the 2007–2009 survey.

A regional breakdown of the census results from different time periods (Table 1) show marked differences in the population development between different Swedish regions. The most marked decrease was found for the areas east of Gotland also including the Hoburg Bank. Marked decreases were also found for the waters north and east of Öland but the magnitude of the decrease was much smaller than for the areas east of Gotland.

On the mainland coast, the Long-tailed Duck is found wintering in the outer parts of the archipelagos from the Kalmarsund in the south to the coasts of Uppland in the north. Numbers here are much smaller than at the offshore banks, but there was no clear trend in the wintering numbers taking the archipelagos as a unit. As a matter of fact, the numbers estimated for the Stockholm archipelago was the highest during the study. It may however be noted that wintering Long-tailed Ducks nearly disappeared from the inshore parts of the south coast of Sweden during the study period.

Different factors have been discussed in relation to the decline such as oil pollution, bycatches in fishing equipment and decreased habitat quality (Larsson & Tydén 2005, Stempniewicz 1994, Mathiasson 1970, Skov et al. 2011, Zydelis et al. 2009) and probably the decline is caused by a combination of different factors. The regional differences in the extent of the decline in Swedish waters with the most marked decline for the waters east of Gotland is however a strong indication that oil pollution is an important factor behind the decline. These areas are close to the main shipping line through the Baltic to St Petersburg and other im-





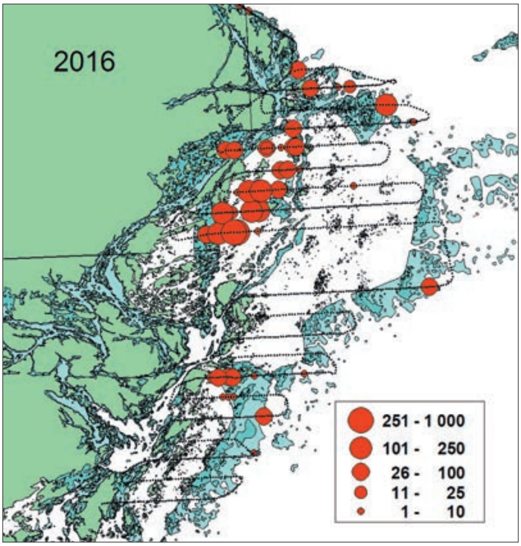


Figure 12. Distribution of wintering Long-tailed Ducks *Clangula hyemalis* based on aerial surveys in the Stockholm archipelago in 2009, 2015 and 2016.
Utbredning för övervintrande alfåglar Clangula hyemalis baserad på flyginventeringar i Stockholms skärgård 2009 2015 och 2016.

portant cities in the eastern part of the Baltic. There is also an extensive but illegal cleaning of the tanks offshore with regular small oil spills at sea.

Acknowledgements

The offshore aerial surveys reported here as well as the Swedish Midwinter Counts are supported by grants from Naturvårdsverket (the Swedish Environmental Protection Agency). The same applies to the aerial surveys in the seventies which were also supported by the Swedish Airforce, which provided free flying time. I am also grateful for the support of the Swedish Coast Guard that allowed duck counters onboard their patrol ships during the surveys in the sixties and seventies.

The aerial surveys could not have been done without the help of a number of skilled observers and I will give my sincere thanks to David Erterius, Martin Granbom, Martin Green, Fredrik Haas, Nils Kjellen, Johan Nilsson, Gabriel Norevik, Roine Strandberg and Mikael Svensson for many observer hours over the open sea

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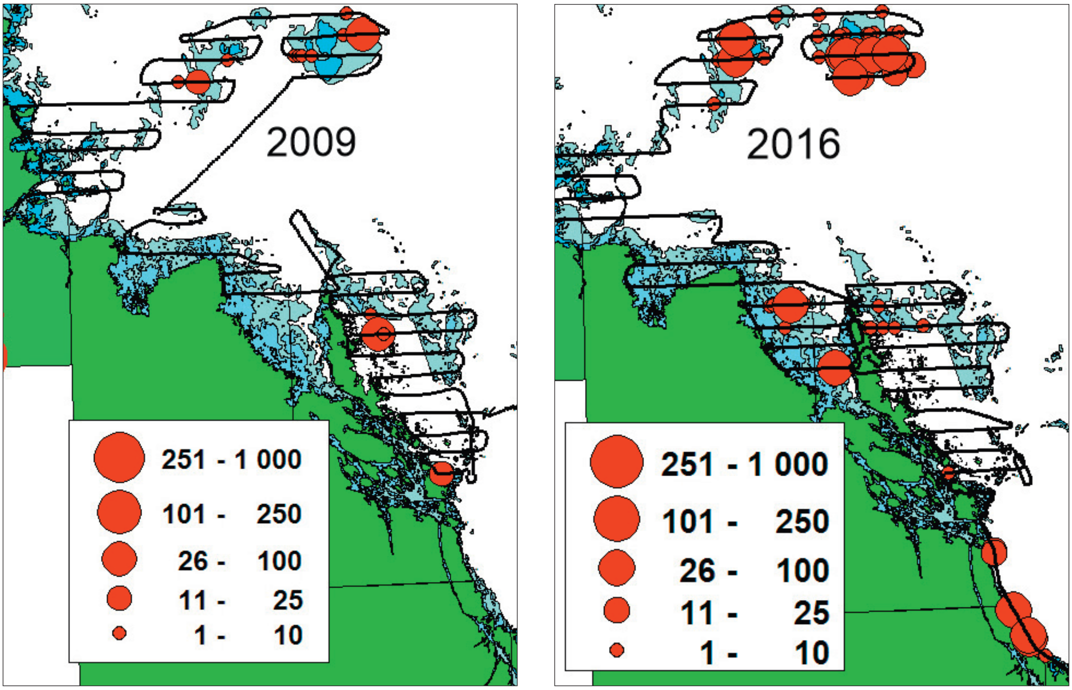

Figure 13. Distribution of wintering Long-tailed Ducks *Clangula hyemalis* based on aerial surveys in the county of Uppsala and the Gävle Bight in 2009 and 2016.
Utbredning för övervintrande alfåglar Clangula hyemalis i baserad på flyginventeringar i Uppsala län och Gävlebukten 2009 och 2016.

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Long-tailed Ducks are counted from aircaraft (this study, upper photo) or from boat or land.
Alfåglar räknas från flygplan (i denna undersökning; övre fotot) eller från båt eller land.

Svensk sammanfattning

Alfågeln är den vanligaste övervintrande andfågeln i de svenska farvattnen, men till följd av dess utbredning långt ute till havs, är den inte så väl täckt vid de regelbundna midvinterinventeringarna av sjöfåglar, som organiserats i Sverige och övriga Europa sedan 1967 (Nilsson 1980, 2012, Nilsson & Haas 2016). Inventering av de övervintrande alfåglarna kräver speciella inventeringar av skärgårdarna och utsjöbankarna.

Under 1965–1978 genomfördes mer än 100 båtinventeringar runt kusterna från kustbevakningens båtar (Nilsson 1972, 1980). Omfattande flyginventeringar genomfördes också i skärgårdarna samt efter de öppna kusterna av Skåne, Öland och Gotland. Däremot var det inte möjligt at inventera utsjöbankar som Hoburgs Bank och Midsjöbankarna, även om dessa var välkända som viktiga lokaler för alfågeln.

I början av 1990-talet, efter de politiska förändringarna i området, kunde för första gången hela Östersjön inventeras med avseende på övervintrande sjöfåglar (Durinck et al. 1994). En ny stor inventering, den s.k. SOWBAS-inventeringen (”Status of Wintering Waterbird Populations in the Baltic Sea”) genomfördes 2007–2011 (Nilsson 2012, Skov et al. 2011). Jämförelser mellan de två totalinventeringarna visade på en mycket markant nedgång av antalet övervintrande alfåglar i Östersjön. För att följa upp situationen organiserades en tredje heltäckande inventering 2016. Dessa sjöfågelinventeringar ingår numera i de indikatorer som utarbetats för Eus marina direktiv för att följa upp situationen i havsområdena.

I denna uppsats analyserar jag resultaten från offshore-inventeringarna i de svenska farvattnen 2016 med fokus på alfågeln. Tidigare resultat från de äldre inventeringarna har sammanfattats av Nilsson (1980), medan resultaten från SOWBAS-inventeringen presenteras av Nilsson (2012). Förutom att presentera de senaste inventeringsresultaten för alfågeln i de svenska farvattnen försöker jag också belysa utvecklingen under de senaste femtio åren mot bakgrund av de äldre inventeringarna.

Material och metoder

Vid inventeringarna 2016 liksom 2007–2011 täcktes de yttre svenska farvattnen och skärgårdsområdena med linjetaxeringar från flygplan. Inventeringslinjer lades ut med 2 eller 4 km mellanrum från kusten ut till ca 30 m djup på sådant sätt att alla potentiella områden för havslevande dykänder kunde täckas. Linjesystemet framgår av översiktskartan i Figur 4 och från de regionala kartorna.

Inventeringarna genomfördes med en högvingad tvåmotorig CESSNA 337-Skymaster. Flyghöjden var ca 70 m och hastigheten vid inventeringarna 180 km/tim. Två observatörer täckte var sin sida av planet. Observationerna registrerades i tre bälten: A ut till 200 m, B 200–500 m och C 500–1000 m. Under planet fanns en död zon så det inre bältet täckte totalt en 320 m bred zon. Navigeringen baserades på fasta ”way-points” vid transekternas början och slut. Dessutom registrerades regelbundet positionerna var 10 sek. på en separat GPS.

Antalet sjöfåglar beräknades för de olika områdena genom att multiplicera vad som räknats inom band A med en faktor på 6,25 resp. 12,5 beroende på om linjerna låg 2 eller 4 km från varandra. I den här presenterade analysen utnyttjades endast observationer från bälte A eftersom endast detta bälte täcktes vid inventeringarna 2007-2011.

Vid inventeringen 2016 täcktes de yttre farvattnen från Falsterbo i söder till Gävlebukten i norr (Figur 2). I skärgårdarna inventerades endast de yttre delarna, medan de inre skärgårdsområdena inventerades 2015 (se Nilsson & Haas 2016). Området inventerades under tio dagar mellan mitten av januari och mitten av mars. Den totala flygtiden var 65 timmar, varav 43 ägnades åt direkta fågelräkningar.

Resultat

Den nationella bilden

Vid den första totalinventeringen av Östersjön 1992/93 (Durinck et al. 1994), beräknades antalet övervintrande alfåglar i de svenska farvattnen till 1,4 miljoner (Figur 1, Tabell 1). Vid SOWBAS-inventeringen 2009 var motsvarande antal 436000. Liknande minskningar konstaterades överallt i Östersjöområdet. Den kalla vintern 2010 var antalet högre, ca 700000 i de svenska farvattnen, men då var de viktiga lokalerna vid Rigabukten och i öster istäckta.

Inventeringen 2016, som liksom 2009 genomfördes under en mild vinter, visade en beräknad summa för de svenska farvattnen på 370000, en fortsatt nedgång. Det är dock omöjligt att avgöra om detta är en del av en allmän minskning eller om alfåglarna ändrat vinterutbredning till viss del, eftersom resultaten från övriga delar av Östersjön ännu inte presenterats.

Under de milda vintrarna noterades övervintrande alfåglar inom hela det inventerade området (Figur 1, se också Nilsson 2012). En mycket betydande del av beståndet var koncentrerat till de tre utsjöbankarna. Hoburgs Bank, Södra och Norra Midsjöbanken. Vid båda tillfällena återfanns ca 70% av alfåglarna på dessa tre bankar. Isvintern 2010 var koncentrationen till bankarna ännu mer markerad, 90% av beståndet låg här.

Regionala mönster

Den regionala fördelningen av de övervintrande alfåglarna vid den senaste inventeringen 2016 redovisas i en serie detaljerade kartor (Figur 3 -13), vilka där så är tillämpligt också visar utbredningen vid tidigare tillfällen.

Som redan nämnts har antalet alfåglar längs den skånska sydkusten minskat sedan starten av inventeringarna på 1960-talet och endast få ses numera i de strandnära vattnen. Flockar konstaterades dock i de yttre områdena. I Hanöbukten har alfåglarna karterats sedan 1960-talet. Utbredningen vid de senaste inventeringarna var densamma som vid de tidiga inventeringarna. Storleksordningen av beståndet höll sig också fram till de senaste inventeringarna, då en minskning noterades.

Vattnen i Kalmarsund och Öland utgör viktiga vinterområden för alfågeln (Figur 5). I Kalmarsund varierar antalet mycket beroende på isförhållandena, medan farvattnen öster om Öland, speciellt i den norra delen hyser större mängder. Grova skattningar från 1970-talet tyder på att antalet alfåglar här kan ha varit så högt som 100 000, medan sentida beräkningar visat på lägre tall, 13000 en mild vinter som 2016 och 61000 under en av isvintrarna.

Betydelsen för Midsjöbankarna och Hoburgs bank som alfågellokal har redan kommenterats. Området utgör tillsammans det absolut viktigaste området för arten med 70% av totalbeståndet under normala vintrar. Tyvärr saknas jämförelse inventeringar från tidigare år. Alfåglarna förekommer spridda över hela bankarna (Figur 6,7).

Gotland utgör också ett viktigt område för alfågeln med flera förekomster runt kusterna. Markanta nedgångar har observerats för detta område, där antalet vid inventeringarna 2009 och 2016 skattades till mellan 11000 och 16000 att jämföras med kanske 200000 1992/ 93 och troligen ännu fler på 1970-talet.

Ostkust skärgårdarna norr om Kalmarsund utnyttjas regelbundet av övervintrande alfåglar. Skärgårdarna i Kalmar, Östergötland och Södermanlands län är mindre djupa än Stockholms skärgård. Hela detta skärgårdsområde inventerades intensivt också under 1970-talet från kustbevakningens båtar och från flyg, varför det finns goda möjligheter att jämföra utbredningen mellan dessa tidiga inventeringar och inventeringarna 2009–2016 (Figur 9–12). En hel del skillnader noterades också i utbredningen mellan perioderna för Stockholms skärgård (Figur 11, 12).

Antalsmässigt förekom inga större skillnader i antalet alfåglar efter kusterna i de sydligare ostkustskärgårdarna fram till och med 2009, men 2016 var antalet i dessa tre län väsentligt lägre än tidigare. I Stockholms län noterades däremot inga skillnader i antalet övervintrande alfåglar mellan de olika perioderna, enbart en del variation. Antalet övervintrare 2016 var det högsta noterade för området.

Diskussion

De här presenterade inventeringarna liksom tidigare publicerade äldre inventeringar (Nilsson 2008, 2012) visar tydligt tillsammans med rapporterna från Östersjöinventeringarna (Durinck et al. 1994, Skov et al. 2011) Sveriges betydelse för de övervintrande alfåglarna i Europa och Östersjön i synnerhet. Totalt återfanns ca 30% av alla Östersjöns alfåglar i de svenska farvattnen, varav 20% på Midsjöbankarna och Hoburgs Bank.

De redovisade internationella inventeringarna liksom de här och tidigare (Nilsson 2012) redovisade svenska inventeringarna visar klart på en markant nedgång i antalet övervintrande alfåglar, en minskning som fortsatt till 2016. Det är emellertid oklart om den senaste minskningen i svenska vatten återspeglar en generell minskning eller endast en omfördelning av utbredningen. Minskningen från 1990-talet återspeglas också i sträckräkningar från Finland (Hario et al. 2009, Kauppinen & Leivo 2008).

Minskningen i antalet övervintrande alfåglar är inte densamma inom hela det svenska övervintringsområdet utan är mest markant för områdena öster om Gotland, men återfinns också utanför Öland. Däremot har endast mindre förändringar eller enbart fluktuationer noterats för kustområden och skärgårdar, där antalet övervintrande alfåglar dock är betydligt lägre. Tyvärr saknas äldre data från de viktiga utsjöbankarna.

Orsakerna till den markanta nedgången är omdiskuterad. Olika faktorer såsom bifångst vid fiske, oljeföroreningar samt minskad kvalité på födosöksområdena har diskuterats (Larsson & Tydén 2005, Stempniewicz 1994, Mathiasson 1970, Skov et al. 2011, Zydelis et al. 2009), men troligen har vi en påverkan av flera olika faktorer. Det faktum att den mest markanta nedgången noterats för farvattnen öster om Gotland ger dock en stark indikation på att faktorer som diffusa oljeutsläpp i den viktiga sjöfartslinjen öster om Gotland kan vara en viktig påverkansfaktor.

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Improving population estimates of Swedish birds using the Breeding Bird Survey fixed routes and correction factors from Finnish line transect surveys

Förbättring av populationsskattningar av svenska fåglar med hjälp av Svensk Fågeltaxerings fasta rutter och korrektionsfaktorer från finska linjetaxeringar

SÖREN SVENSSON

Abstract

I estimated population size of fifty-nine common birds by using the fixed routes of the Swedish Bird Survey (SBS), which give the number of birds per kilometer. I converted this number to density, birds/km^2, using the correction factors for detectability that have been developed for line transects in Finland. I compared the population estimates by this new method with those in a previous account from 2012, in which the estimates of common birds were primarily based on extrapolation of habitat-specific densities from numerous territory mapping plots. There was good agreement for the most abundant species but a clear tendency that the estimates with the new method were higher for many less common ones. As little new density data are being collected, the SBS fixed routes are likely to be the prime source of data for future national population estimates. Although the Finnish correction factors can be used to improve the Swedish estimates for suitable species it is advisable to develop factors specifically adapted to the Swedish counts for application to a wider spectrum of species.

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Introduction

How many pairs of breeding birds are there? The question is asked not only by population and community scientists or by curious laymen. Increasingly, it is asked by governmental bodies at different levels from perspectives of conservation, management or legislation. For example, the Swedish Environmental Protection Agency (SEPA) contracts collection of bird population data, partly for its own use within the country, partly because it is required to report to the European Union for compilation of environmental indicators. One example where Swedish data have contributed is the assessment of how strategic conservation measures within the EU Common Agricultural Policy have affected birds (Gamero et al. 2016). Concern about bird populations has often been a part of the rationale for different international conventions: Bern (habitats), Bonn (migratory species) and Ramsar (wetland) conventions and Convention on Biological Diversity (Nagoya protocol, ratified by Sweden 2016). But there are no firm requirements about reporting national population estimates. Within the European Union the Birds Directive (2009/147/EC) is a core document. Recently an important addition has been enforced. It is now required by all member states to report absolute population size of all wild birds.

There are also non-governmental organizations that need absolute population estimates, for example the International Union for Conservation of Nature for the so called red lists of endangered birds (www.iucnredlist.org). One of several regional lists is the one compiled by BirdLife International (2015) for Europe. In Sweden a national list is published every five years, the most recent one in 2015 (Artdatabanken 2015). Categorization of the bird species into different threat categories requires detailed information on both trends and absolute numbers.

There have been four efforts to estimate the population size of all bird species breeding in Sweden: Ulfstrand & Högstedt (1976), Koskimies (1993), Asbirk et al. (1997) and Ottosson et al. (2012). In Ottosson et al. (2012) the estimates for common species were based on habitat-specific densities multiplied by the national area of each habitat. For less common and rare species other methods were used, primarily a careful search of all literature, both regional and national. For a small number of species with very little information available "educated guesses" were made, for example by comparing with similar species with better known status. The vast majority of habitat-specific density values emanated from territory mapping plots that were surveyed in the period 1970–1990 and apart from more recent surveys of farmland, few estimates of true densities have been made after 1990. Details are given in Ottosson et al. (2012).

In 1996, a new national scheme for monitoring population trends was launched as part of the Swedish Bird Survey (SBS; Svensson 1996, 2000; Lindström et al. 2007). Before 1996 trends were based on territory mapping plots and point counts at sites chosen by volunteers, which were rather non-representative with respect to both geography and habitats. The new scheme was designed to provide data representative for different habitats and geographical regions, and it has been the main instrument for population trend monitoring since then (Green & Lindström 2015). Seven hundred and sixteen fixed survey routes are distributed systematically over all Sweden, twenty-five kilometers apart in both south-north and west-east directions. Each route comprises eight one-kilometer line transects and eight five minute point counts in between each. The routes are surveyed once a year during a three-week period adapted to local phenology to capture as many species as possible, starting on 15 May in the south and on 15 June, or later if necessary, in the northern mountains. A survey of a fixed route always starts at four o'clock in the morning and lasts five to seven hours. All bird individuals heard or seen are recorded, independent of distance, and excluding only downy young, unfledged nestlings, and obvious double counts between adjacent line kilometer sections.

Line transects have long been used in Finland. The Finnish method (Koskimies & Väisänen 1991) differs in three respects from the Swedish fixed routes. First, the count unit in Finland is a pair equivalent (one male, one female, one pair or one brood without parent) instead of an individual bird. Second, observations are recorded separately for two zones, within and beyond twenty-five meters on both sides of the transect line. The zones are called the *main belt* and the *supplementary belt,* together making up the *survey belt*; the latter corresponds to the belt without limits (and hence without defined area) used in the Swedish fixed routes. The proportion of records within twenty-five meters can be used as a measure of lateral detectability as demonstrated in a series of papers (Järvinen & Väisänen 1975, 1976a, 1976b; Järvinen 1976, 1978; Järvinen et al. 1976, 1977). Counts of birds with strong and far-reaching calls will show a lower proportion of records closer than twenty-five meters than birds with weak calls. This makes it possible to calculate species-specific detectability factors that can be used to make the counts of different species comparable. This detectability is called *lateral detectability* as distinguished from *basal detectability* which tries to correct also for birds that are not at all detected, not even close to the transect line. Lateral detectability correction factors were first published by Järvinen & Väisänen (1983) and recently new factors were published by Lehikoinen et al. (2014). Factors correcting for basal detectability have been published by Rajasärkkä (2010). A third difference is walking speed. The line transects of the Swedish fixed routes are walked with a speed of one kilometer per 30–45 minutes, mainly adapted to how difficult the terrain along the route is. The Finnish transects are walked with a speed of 45–60 minutes per kilometer. A Finnish route thus takes more time because the observer is required to estimate the distance to each bird and to put down its position on a map together with habitat information.

It is comparatively easy to monitor national temporal trends of breeding bird species populations. It is sufficient to determine a geographically representative relative measure, an index, which is comparable from year to year. This goal can be achieved, without any knowledge of absolute density, by counting birds with the same standard method every year at a selection of statistically representative sample sites. The fixed routes of the Swedish Bird Survey provide such an index.

Absolute population size is more difficult to determine. There are two alternatives. The first is to determine the absolute numbers in a set of representative sample plots of known size. This will produce density values that, with appropriate precautions, can be extrapolated to the whole country. The second alternative is to use a suitable density index and convert it to density by an experimentally determined correction factor. The Swedish Bird Survey provides such an index for all species, the number of individual birds recorded per kilometer line transect.

The purpose of this paper is twofold. First, I intend to calculate better national estimates than before for a selection of species that I believe are particularly suitable by using the Finnish correction factors (Rajasärkkä 2010, Lehikoinen et al. 2014) to convert the SBS density indices to absolute densities. These new estimates will be compared with those of Ottosson et al. (2012). Second, I provide some ideas on how to develop correction factors that are better adapted to the SBS counts than the ones from Finland and that would allow conversion of SBS indices to densities with greater precision and for a larger number of species.

Methods

To account for the fact that the unit of a Finnish line transect count is a pair equivalent rather than an individual bird, I selected species in which the SBS records are mainly singing males. A singing male represents a pair and if the proportion of singing males among all records is high, a count of individuals becomes a good representation of the number of pairs. A majority of the territorial songbirds in northern Europe belongs to this category. To these passerines I added six woodpeckers and the Cuckoo, assuming that they have territorial systems and calling behaviors similar to those of the passerines. In total, fifty-nine species were included. In sheer numbers, these species represent more than 80% of all breeding pairs in Sweden according to Ottosson et al. (2012). In order to provide further support for the assumption that most fixed route records for these fifty-nine species are pair equivalents, I examined 220,000 five-minute point counts. If most records during a point count were records of a single individual, hence a pair equivalent, most records along a line transect also ought to be pair equivalents. Species with more than 85% such single individual records are marked with an asterisk in Table 1. My assumption is that virtually all fixed route records of the selected fifty-nine species represent pair equivalents and that no correction for this potential bias is necessary.

I used the line transect values given in table A2, pp. 552–556 in Ottosson et al. (2012) as the Swedish sample. These values were based on all fixed routes surveyed during the period 1996–2010 and they represent average values from 4920 eight kilometer surveys and hence a total of 39,360 kilometers of line transect. As these values showed the number of birds per route (eight kilometers), I recalculated them to birds per kilometer. The number of records per kilometer fixed route was first multiplied with the correction factors for lateral detectability in column KK or KE in table 2 in Lehikoinen et al. (2014). As the correction factors differed between northern and southern Finland (higher in the south) and as a large part of Sweden is located south of Finland, I used the values for southern Finland (column KE) for species with mainly a southerly distribution in Sweden; for the other species the value for all Finland (column KK) was used. For correction of basal detectability, I then multiplied with the values of column T in table 1 in Rajasärkkä (2010). This gave an estimate of pairs/km^2. Finally, I multiplied this density estimate with the total area of Sweden, 450 000 km^2.

The nomenclature follows Ottosson et al. (2012) and the English, scientific and Swedish names of birds are found in Table 1. Elsewhere in the paper I use the shorter versions of the English names or, in the figures, the scientific names.

Results

National population estimates based on the line transect counts after multiplication with both the lateral and basal correction factors are shown in Table 1, together with the estimates in Ottosson et al. (2012). Several interesting comparisons can be made from the two datasets.

However, I first compared the abundance rank of the species according to the raw number of birds per kilometer fixed route with the rank after correction for only lateral detectability (Figure 1). The horizontal bars show how much the rank has shifted, been up- or down-graded, after lateral correction. The two most numerous species (Willow Warbler and Chaffinch) and the two least numerous ones (Golden Oriole and Red-breasted Flycatcher) obtain exactly the same position before and after correction. Among the remaining species, only six end up in exactly the same position. However, minor position shifts tell little of interest as they are not independent but mutually affecting each other. And it is likely that the shifts among the least common species are more error-prone than those among the common species. It is therefore the large deviations among more common species that are of greatest interest. The largest shifts to a lower rank are found for Brambling (shifting from position 3 for raw data to position 8 after lateral detectability correction, 5 positions down), Song Thrush (6 to 10, 4 down), Redstart (10 to 20, 10 down), Cuckoo (15 to 39, 24 down), and Black Woodpecker (34 to 43, 9 down). Large shifts in the opposite direction are found in Great Tit (9 to 4, 5 up), Goldcrest (11 to 6, 5 up), Spotted Flycatcher (19 to 11, 8 up), Blue Tit (22 to 13, 9 up), White Wagtail (23 to 12, 11 up), Willow Tit (24 to 18, 6 up), and Treecreeper (35 to 28, 7 up). The number of shifts in the different directions is almost the same, 23 and 26 respectively, with 10 species remaining in the same position.

Table 1. Estimated breeding populations of birds in Sweden according to line transect count data from the Breeding Bird Survey, transformed to national population size by detectability-correction factors developed in Finland (Lehikoinen et al. 2014, Rajasärkkä 2010), versus estimates in Ottosson et al. (2012). An asterisk denotes a species with more than 85% single individual records during five minute point counts (see text).

Beståndsskattningar för fåglar enligt data från Svensk Fågeltaxerings linjetaxeringar, vilka omräknats med faktorer för observerbarhet (Lehikoinen et al. 2014, Rajasärkkä 2010), i förhållande till skattningar i Ottosson et al. (2012). En asterisk anger att arten i 85% av alla fem-minuters punkttaxeringar registreras med bara en individ (se text).

English name	*Scientific name*	Swedish name	Corrected line transects	Ottosson et al. (2012)
Willow Warbler	*Phylloscopus trochilus*	Lövsångare	12 000 000	13 000 000
Common Chaffinch	*Fringilla coelebs*	Bofink	9 200 000	8 400 000
European Robin	*Erithacus rubecula*	Rödhake	3 300 000	3 800 000
Great Tit	*Parus major*	Talgoxe	3 300 000	2 600 000
Goldcrest	*Regulus regulus*	Kungsfågel	2 700 000	3 000 000
Tree Pipit	*Anthus trivialis*	Trädpiplärka	2 600 000	2 400 000
Brambling	*Fringilla montifringilla*	Bergfink	2 200 000	2 100 000
Common Blackbird	*Turdus merula*	Koltrast	2 200 000	1 800 000
Redwing	*Turdus iliacus*	Rödvingetrast	1 800 000	1 100 000
Spotted Flycatcher	*Muscicapa striata*	Grå flugsnappare	1 800 000	1 500 000
Song Thrush	*Turdus philomelos*	Taltrast	1 500 000	1 900 000
Eurasian Blue Tit	*Cyanistes caeruleus*	Blåmes	1 400 000	700 000
White Wagtail	*Motacilla alba*	Sädesärla	1 400 000	410 000
Yellowhammer	*Emberiza citrinella*	Gulsparv	1 200 000	900 000
Eurasian Blackcap	*Sylvia atricapilla**	Svarthätta	1 100 000	1 200 000
Garden Warbler	*Sylvia borin**	Trädgårdssångare	1 000 000	1 200 000
European Pied Flycatcher	*Ficedula hypoleuca*	Svartvit flugsnappare	980 000	1 400 000
Willow Tit	*Poecile montanus*	Talltita	968 852	800 000
European Greenfinch	*Chloris chloris*	Grönfink	950 000	660 000
Eurasian Wren	*Troglodytes troglodytes*	Gärdsmyg	740 000	500 000
Common Redstart	*Phoenicurus phoenicurus**	Rödstjärt	700 000	900 000
Whinchat	*Saxicola rubetra*	Buskskvätta	670 000	250 000
Gr. Spotted Woodpecker	*Dendrocopos major**	Större hackspett	630 000	210 000
Common Whitethroat	*Sylvia communis*	Törnsångare	600 000	250 000
Dunnock	*Prunella modularis**	Järnsparv	570 000	630 000
European Crested Tit	*Lophophanes cristatus*	Tofsmes	530 000	400 000
Eurasian Treecreeper	*Certhia familiaris**	Trädkrypare	420 000	750 000
Common Reed Bunting	*Emberiza schoeniclus*	Sävsparv	400 000	400 000
Coal Tit	*Periparus ater**	Svartmes	390 000	410 000
Lesser Whitethroat	*Sylvia curruca**	Ärtsångare	330 000	250 000
Wood Warbler	*Phylloscopus sibilatrix*	Grönsångare	300 000	220 000
Mistle Thrush	*Turdus viscivorus**	Dubbeltrast	280 000	330 000
Common Chiffchaff	*Phylloscopus collybita**	Gransångare	200 000	190 000
Red-backed Shrike	*Lanius collurio**	Törnskata	190 000	44 000
Rustic Bunting	*Emberiza rustica**	Videsparv	180 000	40 000
Bluethroat	*Luscinia svecica**	Blåhake	150 000	230 000
Icterine Warbler	*Hippolais icterina**	Härmsångare	130 000	50 000
Marsh Warbler	*Acrocephalus palustris*	Kärrsångare	130 000	24 000
Common Cuckoo	*Cuculus canorus**	Gök	94 000	78 000
Sedge Warbler	*Acrocephalus schoenobaenus*	Sävsångare	76 000	100 000

Thrush Nightingale	*Luscinia luscinia*	Näktergal	72 000	37 000
Grey-headed Chickadee	*Poecile cinctus*	Lappmes	71 000	54 000
Black Woodpecker	*Dryocopus martius**	Spillkråka	59 000	29 000
Snow Bunting	*Plectrophenax nivalis*	Snösparv	57 000	26 000
Eurasian Three-toed Woodpecker	*Picoides tridactylus**	Tretåig hackspett	55 000	11 000
Eurasian Reed Warbler	*Acrocephalus scirpaceus*	Rörsångare	53 000	290 000
Common Rosefinch	*Carpodacus erythrinus**	Rosenfink	33 000	17 000
Hawfinch	*Coccothraustes coccothraustes*	Stenknäck	33 000	17 000
Ortolan Bunting	*Emberiza hortulana*	Ortolansparv	27 000	6 300
Eurasian Wryneck	*Jynx torquilla*	Göktyta	26 000	25 000
Pine Grosbeak	*Pinicola enucleator*	Tallbit	24 000	10 000
Le. Spotted Woodpecker	*Dendrocopus minor**	Mindre hackspett	20 000	7 000
Woodlark	*Lulula arborea**	Trädlärka	19 000	15 000
Ring Ouzel	*Turdus torquatus*	Ringtrast	14 000	6 200
Great Grey Shrike	*Lanius excubitor**	Varfågel	11 000	6 000
Grey-headed Woodpecker	*Picus canus**	Gråspett	7 400	1 900
Grashopper Warbler	*Locustella naevia**	Gräshoppsångare	6 700	4 600
Red-breasted Flycatcher	*Ficedula parva**	Mi. flugsnappare	3 900	1 100
Eurasian Golden Oriole	*Oriolus oriolus**	Sommargylling	560	120

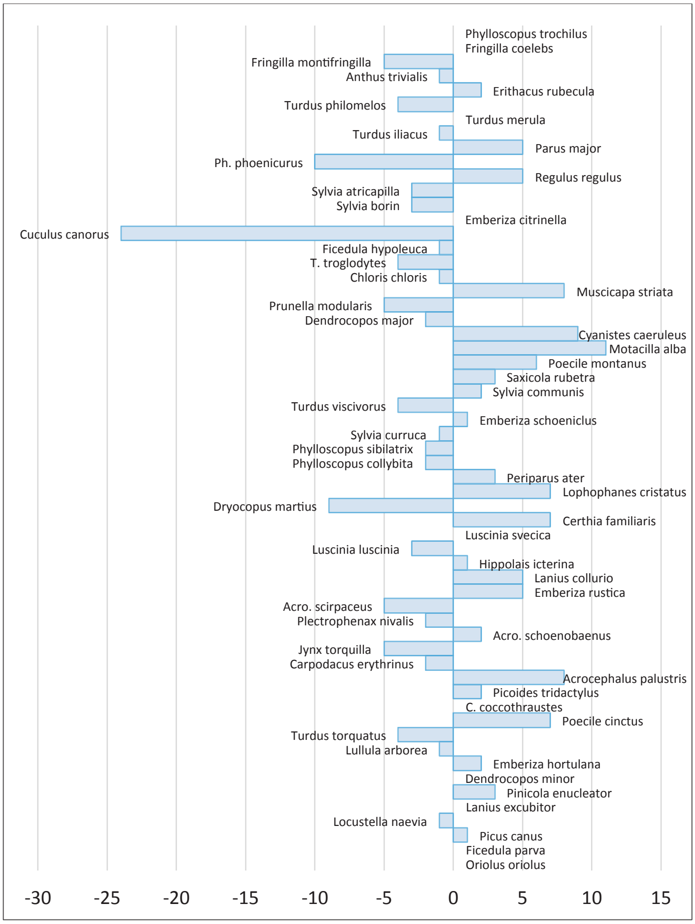

Figure 1. The shift of abundance rank from the rank a species had according to the raw data from the Swedish Bird Survey (birds/km) to the rank it obtained after multiplication with Finnish correction factors for lateral detectability. For example, the Cuckoo *Cuculus canorus* with far-reaching calls has moved 24 steps down in rank and the modest Spotted Flycatcher *Muscicapa striata* 8 steps up in rank. The species that did not shift rank have no bars. In the diagram the most common species at top, the least common at bottom.
Förändringen i rangordning från den position en art hade enligt rådata från Svensk Fågeltaxering (individer/km) till den position den fick efter multiplikation med finska korrektionsfaktorer för lateral observerbarhet. Till exempel flyttas göken med sin ljudliga stämma 24 steg ner i rangordning medan den mera diskreta grå flugsnapparen flyttas upp 8 positioner. Arter som inte bytte position saknar stapel. I diagrammet står de vanligaste arterna överst och de fåtaligaste nederst.

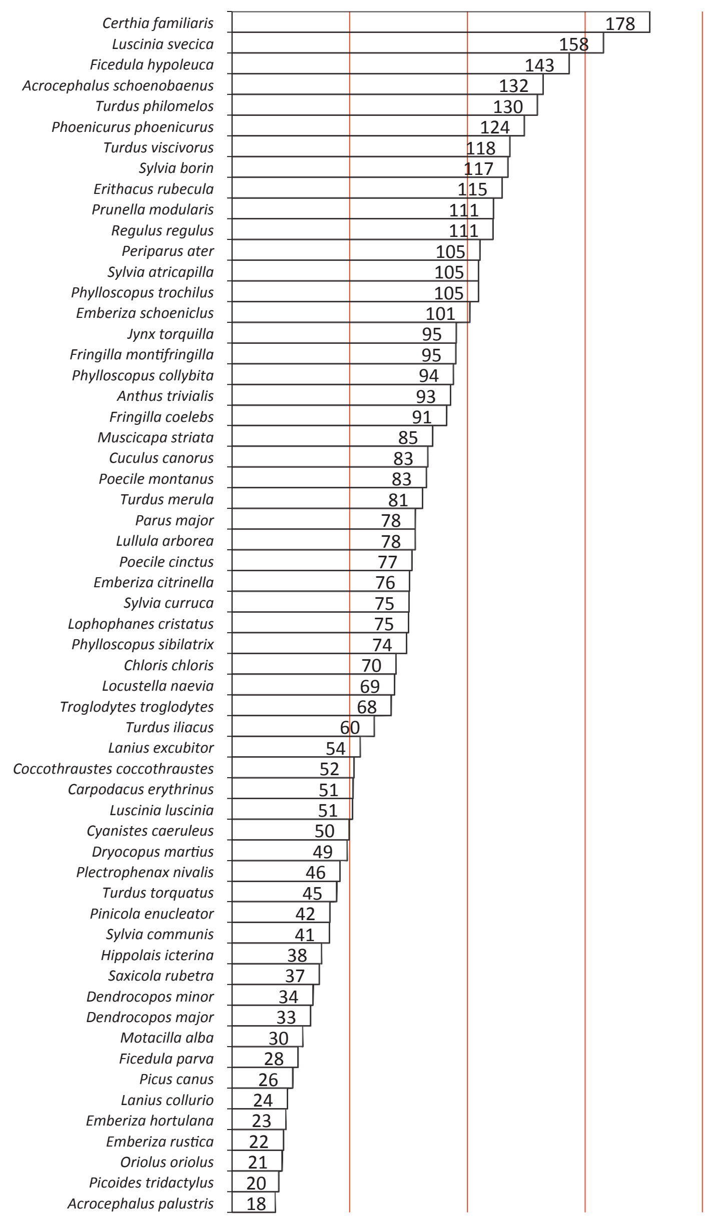

Figure 2. National breeding population estimates for Sweden in Ottosson et al. (2012) as a percentage of the new estimates based on detectability-corrected line transect counts of the Swedish Bird Survey. For example, the estimate of Treecreeper *Certhia familaris* was 750,000 pairs in Ottosson et al. (2012) and 420,000 pairs with the detectability-corrected method (this study), where the first estimate is 178% of the latter. Reed Warbler *Acrocephalus scirpaceus* (549%) is omitted from the diagram.

Beståndsuppskattningarna för Sverige enligt Ottosson et al. (2012) i procent av de nya skattningarna baserade på linjetaxeringarna i Svensk Fågeltaxering efter korrigering medfinska detektabilitetsfaktorer (denna studie). Till exempel hade trädkryparen en beståndsskattning på 750.000 par i Ottosson et al. (2012) och 420.000 par efter korrigering för detektabilitet (denna studie), och det förstnämnda värdet är 178% av det senare. Rörsångaren (549%) har uteslutits från diagrammet.

The national population estimate in Ottosson et al. (2012) is higher than the estimate based on corrected line transect values in sixteen species and lower in the remaining 43 species (Table 1, Figure 2). In one species, the Reed Warbler, the estimate in Ottosson et al. (2012) is very much higher (290,000 vs. 53,000 pairs, 543% higher). A related species, the Sedge Warbler, has a much lower value, only 132% (97,000 vs 71,000). Between these species fall the Treecreeper, Bluethroat and Pied Flycatcher (about 1.5 times higher). There are 40 species for which the estimate in Ottosson et al. (2012) falls in the region 50%–200% of the corrected line transect estimate, i.e. half or twice in

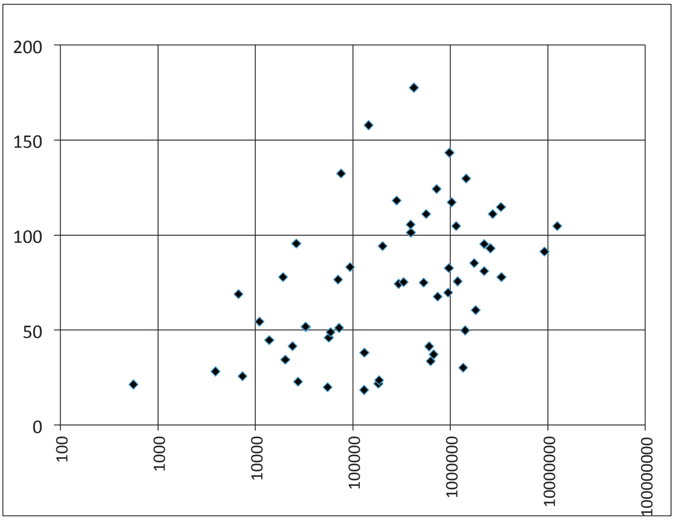

Figure 3. The percentage difference between estimates in Ottosson et al. (2012) and estimates based on detectability-corrected line transect data from the Swedish Bird Survey (y axis) in relation to total abundance (x axis). *Acrocephalus scirpaceus* (549%) was excluded from the diagram.
Procentuella skillnaden mellan skattningarna I Ottosson et al. (2012) och skattningarna baserade på de korrigerade värdena från Svensk Fågeltaxering (y-axeln) i förhållande till artens beståndsstorlek (x-axeln). Rörsångaren (549%) är ej medtagen i diagrammet.

numbers. Within the narrow range of 75% to 125% there are 25 species. For as many as 18 species, though, the estimate in the Ottosson et al. (2012) is below 50% of the corrected values calculated here. Among all 59 species, 43 (73%) had lower estimates in Ottosson et al. (2012). Thus, there is a strong bias towards underestimates in Ottosson et al. (2012) compared to values based on the SBS line transect values corrected for detectability.

The frequency and size of these underestimates differ depending on how common a species is (Table 1, Figure 3). For species with less than 100,000 pairs, 19 out of 21 species (90%) are lower, and for more common birds the corresponding figures are: 100,000–1,000,000 pairs, 14 of 22 species (64%); > 1,000,000 pairs, 10 of 16 species (62%).

For the fifty-nine species together the Ottosson et al. (2012) estimate is 56 million pairs, whereas that based on detectability-corrected line transects is 60 million pairs. The small difference between these totals is mainly due to the fact that the estimates of the two most abundant species are very similar.

Discussion

There is no doubt about the demand of quantitative information about bird populations. As mentioned in the introduction Sweden is required to report that information to EU according to the Birds Directive. The question is only how precise it must be. But for one purpose, exact numerical values are required, namely for assessing the threat level of a species in the IUCN Red list of threatened species. For example, Sweden applies a limit of less than 500 pairs for a non-declining and 5000 pairs for a declining species to be included among the threatened species (i.e. at least "vulnerable"). In Sweden the red list has a high degree of authority as it is adopted, on behalf of the government, as an official document by the SEPA and also used as such by county administrative boards and municipal governments.

Most species included in this paper are rather abundant and not candidates for red-listing on the basis of their population size only. But if accurate correction factors for all species in Sweden could be developed, the SBS line transects would be an excellent instrument for precise estimates of population size of many species, even quite rare ones.

Do the Finnish correction factors work in Sweden?

There are several difficulties involved in using the line transect counts as so many confounding factors are involved, for example habitat, geographical region, date, community and species density, weather, phenology, and observer skill. Standardization can only partly compensate for these possible biases. The potential sources of error in bird counts have been discussed numerous times during the history of bird surveys, for example by Enemar (1959). The Finnish correction factors are of course liable to the same difficulties, and they were discussed and analysed by Järvinen and Väisänen in their many papers referred to above. Clearly, errors, variation and bias may emanate from both the

Swedish line transect counts and from the Finnish correction factors.

One important question is whether the Finnish correction factors can be applied in all of Sweden. Sweden and Finland differ in some respects. For example, a large part of Sweden is located south of Finland and a larger proportion of northern Sweden is montane. But in spite of these differences the features of the predominant vegetation type in both countries, commercial forest, are similar. This means that there is only little a priori expectation that the use the Finnish corrections should give drastically misleading results.

The correction factors in Lehikoinen et al. (2014) are different between southern and northern Finland. The average for southern Finland was 4.6 and for northern Finland 5.4 (calculated for the 42 species with values for both parts of Finland). It is difficult to judge how different the estimates would be if correction factors existed for southern Sweden. I made the calculations both with and without the separate correction factors for southern Finland, but the differences were marginal. This may indicate that special factors for southern Sweden would not make much difference. However, as demonstrated by Järvinen & Väisänen (1983) total density of the community affects the species estimates by affecting detectability. As total density is lower in the north and higher in the south, northern species tend to be underestimated and southern species overestimated. In Finland, one therefore also corrects for this latitudinal effect (Virkkala & Lehikoinen 2014). This correction cannot be applied in Sweden because it requires main belt densities which are not available. Hence, this is one argument for developing specifically Swedish conversion factors.

The correction factors for basal detectability (Rajasärkkä 2010) differed very little among species (1.4-1.7, with an average value of 1.53). This moderate variation is what one would expect since the large variation in lateral detectability has already been removed. Ideally, the correction factor should be determined by carrying out line transects in plots with well-known densities of all relevant species. Two such studies have been carried out in very small plots with uncertain results (Järvinen et al. 1978a,b). A third study made by Tiainen et al. (1980) was larger in scope and intended to better represent the mixture and proportion of different habitats normally included in a Finnish line transect. In this case the basal detectability was 65%, which corresponds well with the value of about 1.5 for correction of basal detectability (Rajasärkkä 2010). A single visit in a territory mapping plot is approximately comparable in efficiency to a line transect in such a plot. For example, Enemar (1959) found that single visits showed an average detectability of 60% for 15 species, and a detectability of 71% for the four species with the largest samples. This also suggests a multiplication factor of about 1.5 for basal detectability.

Shifts in abundance rank

The shifts of position in the abundance sequence after lateral correction in Figure 1 are largely the expected ones, namely that species with far-reaching calls and hence a large proportion of records beyond twenty-five meters get a lower rank, whereas secretive and less far-sounding species get a higher rank. Obvious examples are Songtrush, Redstart, Cuckoo and Black Woodpecker (downgraded) and Treecreeper, White Wagtail, Blue Tit, Spotted Flycatcher, and Goldcrest (upgraded). However, it is meaningless to discuss details in the movements of the species along the relative abundance gradient because the ranks depend on each other. For the purpose of this paper I simply accept that the lateral corrections removed most of the differences in lateral detectability.

Note, however, that a large shift in abundance rank does not always imply a large shift in abundance estimate. For example, the Reed Warbler moved only from position 41 to 46 after correction which may seem little, but in number of pairs from 290,000 in Ottosson et al. (2012) to 53,000 based on corrected line transects. This difference is beyond easily explained errors. It is unlikely that the correction factor for lateral detectability is remarkably wrong as it lies in the same range as for a number of warblers. A possible explanation may rather be that the fixed route counts severely underestimate the Reed Warbler because observers walk to the side of the very wet reed habitats thus obtaining a much too low count. The more terrestrial Sedge Warbler is less likely to be unintentionally avoided, but also because it has prominent display flight that can be observed at considerable distance.

The Cuckoo showed a pattern opposite to that of the Reed Warbler. It shifted rank from 14 to 39 after lateral correction, a large shift, but very little in numerical estimates, only from 78,000 to 100,000. This is well within the acceptable estimation error and possibly also within normal between-year population size fluctuation.

Why different estimates

Why are a majority of the estimates by Ottosson et al. (2012) lower than the corrected line transect estimates (Figure 3)? And why is this tendency particularly evident for species with less than one hundred thousand pairs, for which ninety percent show lower estimates? For the two most abundant species there is almost no difference at all. It is expected that the difference between the two methods should increase with declining species abundance as both the amount of density data and sample size for calculation of correction factors decline. However, it was not expected that so many of the less abundant species were to obtain so consistently lower estimates.

The explanation for the agreement between the estimates by the two methods for the commonest species is most probably the fact that very representative density data were available to Ottosson et al. (2012). Most of the abundant species are distributed over large parts of Sweden and in several different habitats, and high quality density estimates from many plots representing all habitats and all parts of the country were available; good trend data for most of the species were also available, so it was possible to adjust the density estimates accordingly for the few species that had changed much in recent years. As the amount of density estimates became scarce with decreasing abundance and distribution of the species the estimates in Ottosson et al. (2012) are expected to turn less reliable.

Another explanation for why estimates in Ottosson et al. (2012) tend to be conservative relative the corrected line transect estimates for less common species could be that the factors for lateral detectability correction are biased in relation to bird abundance. But there is absolutely no deviation from zero slope when the KK values in Lehikoinen et al. (2014) are correlated with population size in Finland.

A likely explanation for many low estimates in Ottosson et al. (2012) is that the authors were deliberately conservative, and more so the less common the species is. The less common a species, the fewer reliable habitat-specific density estimates were available for the population size calculations and the more they had to trust their general opinion about which density values to apply. They were probably affected by a general caution not to overestimate population size. The Precautionary Principle has not been a silent part of the EU or national legislation but has had a profound influence and has permeated thinking and attitudes in nature conservation; best to be on the safe side in the face of possible threats to a species. Additionally, to be conservative is probably an inherent psychological property among ornithologists; for example, when judging the size of a flock or reporting the number of birds after a sweep with the binoculars, one tends to say “at least one hundred” rather than “less than one thousand”.

A comparable study in the UK (Newson et al. 2008) arrived at exactly the same pattern as I have found. They compared population estimates based on habitat-specific densities multiplied with habitat area and detectability-corrected line transect counts. One study was based on the plots of the Common Birds Census (CBC). The CBC used plots where the number of breeding pairs was determined by mapping their territories during several visits distributed over the entire breeding season. The total UK population estimates were then obtained by extrapolation of the densities in relation to the extent of each habitat. The other study was based on the line transects of the British Breeding Bird Survey (BBS). In the BBS the observers counted all birds along a 2 km long transect, one in each of 2500 sites distributed in a stratified random manner all over the UK. The counts were of individuals, not of pairs, and the count was repeated twice a year. The early count was used for residents and the late count for migrants. The distance to all birds was determined with a precision of 0–25, 25–100 and beyond 100 meters. The population estimates were then calculated with a standard program for distance sampling (DISTANCE 5.0) and divided by two to transform number of individuals to number of pairs. Hence, BBS is similar to the Swedish fixed route counts in counting individuals, not pairs, and similar to the Finnish line transects in making distance estimates (0–25 and beyond 25 meters in Finland). Newson et al. (2008) found that for the most common species the population estimates from CBC and BBS data were similar. For 28 species with more than one million pairs according to CBC, the BBS estimates were almost equally distributed about the line of equivalence (13 higher, 15 lower). For species with less than one million pairs the BBS estimates were most often higher than the CBC estimates, and for the very least common species, those with less than one hundred thousand UK pairs according to the CBC estimate, 24 of 29 had higher BBS estimates. Although the line transect counts were corrected by different methods, the differences were the same: the corrected line transects produced higher estimates than habitat-specific densities. Newson et al. (2008) could not explain why the line transects tended to produce higher estimates than the mapping plots, and concluded that there is no way of telling which of the estimates (CBC or BBS) are the correct ones. As a remedy they suggest "to carefully design independent surveys for carefully chosen species that would provide the most reliable baseline for comparison."

A few species as examples

Even taking all the various effects and suggested explanations into consideration, some of the differences between the estimates in Ottosson et al. (2012) and the corrected SBS counts are remarkable. For example, the corrected estimates for Red-backed Shrike and Rustic Bunting are five times higher, about 200,000 versus 40,000 pairs, than those of Ottosson et al. (2012) (Table 1). Similarly, the number of Marsh Warblers is five times higher (more than 100,000 versus 20,000 pairs). Interestingly, the estimates for all three woodpeckers also increased dramatically after correction: Three-toed Woodpecker (five times), Lesser Spotted Woodpecker (three times) and Grey-headed Woodpecker (four times). Being one of the authors of Ottosson et al. (2012), I remember that we had animated discussion about these species. We may have been influenced by the fact that three of the woodpeckers were red-listed in the NT category on the basis of the A criterion (population decline), and therefore unknowingly preferred not to present an estimate that was substantially higher than previously.

The Ortolan Bunting was estimated at 6,000 pairs by Ottvall et al. (2007) and this estimate was accepted by Ottosson et al. (2012). The corrected line transect estimate is much higher, 27,000 pairs. The method used by Ottvall et al. (2007) was similar to the one used here by me in the sense that it was based on the number of birds observed along the lines of the fixed routes of the SBS. But the detectability correction was different, namely an assumption that 75% of the pairs were detected within 400 m on both sides of the transect. Lehikoinen et al. (2014) reported 24% within the 25+25 meters (the main belt; n= 1431 pairs) and Rajasärkkä (2010) a corresponding value of 15% (n=93 birds). The former estimate gives a KK factor of 5,0 in southern and 5.8 in northern Finland, hence, if applied on the Swedish routes, a population estimate of 20,000 to 30,000 pairs. One cannot exclude that the Finnish correction factors are erroneous for some species. On the other hand, is it realistic that 75% of all Ortolan bunting pairs within 400 meters from the observer will be recorded during an average line transect by an average observer? Perhaps a more realistic proportion is 25%? If so the estimate increases to almost 19,000 pairs, at least approaching the other estimate.

Several of the species in these examples underwent considerable population change during the period 1996–2010. The number of birds per kilometer SBS fixed route that have been used in this paper refers to the average value for that period, and for rapidly declining or increasing species the corresponding value for 2008, the datum year for Ottosson et al. (2012), must have been different. For example, the Rustic Bunting and Ortolan Bunting declined with 70–80 percent between 1996 and 2010 and the Grey-headed Woodpecker increase with about 400 percent according to the SBS indices. Hence, care must be exercised when considering the differences observed between the two estimation methods. Large changes in numbers before 2008 were often accounted for in the estimates by Ottosson et al. (2012). The fixed route data from the same source have not been manipulated in any way before use in this paper. Some of the differences between the two methods may therefore be explained by large population changes. And if such population changes have continued after 2010 the corrected fixed route estimates may be still less valid to-day. New estimates based on the fixed routes for the period after 2010 could have been made. I have not done that because my ambition was to compared data from about the same period as used by Ottosson et al. (2012).

Improved estimates of Swedish bird populations: ideas about what should be done

It is difficult to guess how long it will take before someone takes the initiative to seriously revise the population estimates in Ottosson et al. (2012). Certain is, however, that for such a revision to be a substantial improvement, new data and new methods must be used. The existing density data (those from territory mapping plots) have been exhausted and are often very old, having been collected in the 1970s to 1990s (pp. 561–567 in Ottosson et al. 2012). In addition, there is no or little present collection of new habitat-specific densities. So even if future habitat maps, for example satellite images, aerial photographs and different scans of the vegetation, will be both sufficiently detailed and possible to interpret in such a way that the habitats are relevant for bird population estimates, it will not be enough.

Alternative 1: A new scheme for direct determination of absolute densities

The way to do this is obvious. In principle, a network of representative plots must be established all over the country. The most efficient strategy is likely to use stratified random or systematic sampling with more plots in heterogeneous parts of the country and fewer in homogeneous parts. If feasible, the sampling should connect with existing schemes, preferably the joint system of the Breeding Bird Survey and the National Inventory of Landscapes in Sweden (NILS; Ståhl et al. 2011). This would, in addition, potentially provide valuable habitat-specific bird densities for more general extrapolation. The count method must be some form of mapping of territories and, for some species, nests or other breeding cues. The number of visits in a plot will vary, from few in simple, species-poor habitats with a brief and contracted breeding season to several in species-rich habitats with a long collective breeding season. How many years that will be needed for sufficient amount of data to accumulate depends of course on the number of plots that can be surveyed each year, the number of species one aims to cover, and the precision required. This can be calculated as we already know sufficient about both normal densities and normal population variation. Whatever the result of the calculation, such a scheme will be demanding because the mapping method itself is demanding. And as the sample plots must be widely distributed much time will have to be spent travelling. Manpower will be a severely limiting factor even if sufficient funds were made available. And if a high degree of precision is required, a design along the lines described is, although necessary, probably not feasible.

Alternative 2: Application of transformed point and line transect data to estimate absolute numbers

An enormous number of birds have been counted each year for more than twenty years at points and along lines of the Breeding Bird Survey. Indeed, this program has grown and expanded to become the major instrument for trend monitoring of Swedish birds. It is of course tempting to use those data not only for that purpose but also for population size estimates. This paper is the first effort to do so for a large number of species. The general principle for transformation of relative counts to absolute numbers were laid out in Ottosson et al. (2012) and an equation with the relevant species-specific variables was given: maximum distance for a bird to be recorded, and the probability of recording a male and a female, respectively, within that distance. But the values to put into the equation are not known yet.

As it is not yet established to what extent Finnish conversion factors are applicable in Sweden, it is advisable to develop new factors. To obtain Swedish correction factors the simplest and most realistic way is to establish a number of large plots where population size is determined carefully and completely. Line transects and point counts should then be made in these plots by a variety of people in order to determine average correction factors (sensu Tiainen et al. 1980). Factors that are determined in this way will include both the lateral and basal corrections. A rather small number of such plots, easy to reach but covering all essential habitats, should be established and maintained in a few representative parts of the country. I strongly believe that this is the most efficient way to obtain reliable conversion factors for the point counts and line transects of the Swedish Breeding Bird Survey. An alternative way would be to locate a larger number of smaller mapping plots in connection with a representative sample of fixed routes. Then data from several years of point and line counts would already be available and correction factors would be obtained directly. And if one assumes that the correction factors are independent of details of geography, one could locate the test plots where people, especially volunteers of the same kind as those doing the fixed route counts, are easily available, presuming that all major parts of the country become represented. Both approaches can of course be combined.

Acknowledgement

Aleksi Lehikoinen informed me about the correction factors for basal detectability. Johan Elmberg and Raimo Virkkala suggested numerous improvements as referees. Without the dedicated volunteers who made the counts along the many hundreds of routes in Finland and Sweden, the national bird surveys as well as this study would not have been possible. My warm thanks go to all of them.

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Sammanfattning

Tillförlitliga kunskaper om fåglarnas antal behövs inom både internationell och nationell naturvård. Exempelvis har Sverige inom ramen för fågeldirektivet skyldighet att till EU rapportera beståndsstorleken för alla vilda fåglar. Kännedom om beståndsstorleken behövs också för att kunna placera arter i rätt hotkategori vid den så kallade rödlistningen.

I denna uppsats försöker jag förbättra och kontrollera vissa av de skattningar av de svenska fågelbeståndens storlek som gjordes i boken *Fåglarna i Sverige – antal och förekomst* (Ottosson m.fl. 2012). I nämnda arbete skedde skattningarna för allmänna arter i de flesta fall med hjälp av biotopspecifika tätheter som multiplicerades med respektive biotops areal. Täthetsuppgifterna hämtades från allehanda källor, men för vanliga arter till största delen från de hundratals provytor som inventerades med revirkartering inom Svensk häckfågeltaxering fram till början av 1990-talet. Sådana inventeringar sker numera i mycket ringa omfattning varför uppdaterade och aktuella täthetsskattningar för olika biotoper kommer att saknas för framtida beståndsskattningar.

Jag räknar med att den främsta källan i stället kommer att vara de fasta standardrutterna inom Svensk fågeltaxering. Det finns 716 rutter jämnt spridda över hela landet. Varje rutt är åtta kilometer lång och består av åtta delsträckor om vardera en kilometer och åtta punkträkningar om fem minuter vardera mellan kilometersträckorna. Längs kilometersträckorna räknas alla hörda och sedda fåglar, s.k. linjetaxering. Av flera orsaker kan antalet fåglar som registreras under en sådan linjetaxering inte användas direkt för att beräkna tätheter av antal par. En orsak är att fåglarna registreras oberoende av avståndet, alltså inte inom en given areal. En annan orsak är att vi inte vet hur stor andel av de fåglar som finns som verkligen upptäcks, något som varierar mellan olika arter. En tredje orsak är att fåglarna registreras oberoende av kön, vilket innebär att man inte vet hur många par som de registrerade fåglarna representerar, vilket man givetvis måste veta om man vill beräkna antal par per arealenhet. Det behövs således omräkningsfaktorer, även kallade korrektionsfaktorer, för att använda data från standardrutterna. Sådana faktorer saknas för svenskt vidkommande.

I Finland har man använt linjetaxeringar under lång tid (inventeringsmetodiken beskrivs i Koskimies & Väisänen 1991) och där har man tagit fram omräkningsfaktorer. Dessa faktorer kan dock inte utan vidare användas för svenska standardrutter eftersom metoderna skiljer sig i tre avseenden. För det första har man i Finland som räkningsenhet ”parekvivalent” (en hane, en hona, ett par, en kull utan förälder) i stället för en individ som på standardrutterna. För det andra registreras fåglarna inom två zoner, inom och utanför 25 meter från linjen. Detta gör det möjligt att beräkna det som kallas *lateral upptäckbarhet (detektabilitet)*. Denna skiljer sig som sagt kraftigt mellan olika arter. Exempelvis är andelen gökar som registreras inom 25 meter lågt i förhållande till andelen kungsfåglar inom samma avstånd. Genom att multiplicera antalet fåglar per kilometer med korrektionsfaktorn för lateral upptäckbarhet blir registreringarna för alla arter jämförbara om än de fortfarande bara är relativa. Men med ytterligare en korrigering kan absoluta täthetsvärden erhållas, nämligen med korrigering för *basal upptäckbarhet*, dvs. korrigering för fåglar som inte upptäcks alls, inte ens alldeles nära linjen. Korrektionsfaktorer för lateral upptäckbarhet har publicerats av Järvinen & Väisänen (1983), senare uppdaterade av Lehikoinen (2014) och det är de senare jag använder i denna uppsats. Korrektionsfaktorer för basal upptäckbarhet har publicerats av Rajasärkkä (2010).

Jag har utgått från de värden på antal fåglar som i genomsnitt observerats på standardrutternas linjetaxeringar åren 1996–2010. Dessa värden finns redovisade i tabell A2 på sidorna 552–556 i Ottosson m.fl. 2012). Genom att dividera med längden av en standardrutt, åtta kilometer, har jag fått antalet fåglar som registrerats per kilometer. Därefter har jag multiplicerat med de finska korrektionsfaktorerna för lateral och basal upptäckbarhet och på så sätt fått medelvärden för de olika arternas tätheter (per kvadratkilometer) i Sverige. Dessa medelvärden har därefter multiplicerats med Sveriges areal, 450.000 kvadratkilometer. De på detta sätt erhålla totalvärdena motsvarar dock inte det verkliga antalet par eftersom vi i Sverige räknar individuella fåglar längs standardrutterna medan de finska korrektionsfaktorerna är baserade på räkning av par. Det är för närvarande inte känt på vilket sätt antalet individer motsvarar antalet par.

Jag har därför begränsat analysen till arter för vilka jag anser att majoriteten av registreringarna längs standardrutterna bör motsvara antal par, nämligen för många av de vanliga revirhävdande tättingarna för vilka en sjungande hane kan anses motsvara ett par och för vilka andelen registreringar av annat slag är lågt. Jag har valt 59 arter, de flesta tättingar, men därutöver gök och sex hackspettar. Dessa femtionio arter svarar tillsammans för ungefär 80 procent av alla fåglar i Sverige. Resultatet är sammanfattat i Tabell 1, som inkluderar en jämförelse med skattningarna i Ottosson m.fl. (2012).

För att visa hur de olika fågelarterna påverkas efter korrigering för enbart lateral upptäckbarhet, som är den korrigering där korrektionsfaktorn varierar mest mellan olika arter, visar jag i Figur 1 hur deras rang eller ordningstal förändras. I figuren är arterna ordnade från den vanligaste (överst) till den fåtaligaste enligt de okorrigerade värdena från standardrutternas linjetaxeringar, dvs. enligt nyssnämnda tabell A2 i Ottosson m.fl. (2012). Efter korrigering för arternas olika laterala upptäckbarhet kommer exempelvis göken att förflyttas från rang 15 till rang 39, en förflyttning neråt på hela 24 positioner. Detta är helt naturligt med tanke på hur antalet gökar överskattas på grund av de hörs så långt. Med den tystlåtna grå flugsnapparen är det tvärtom. Den flyttas upp 8 positioner i talrikhetsskalan i jämförelse med sin skenbara antalsrankning längs standardrutterna. Att de två vanligaste och de två fåtaligaste arterna inte drabbas av några ändringar är inte förvånande med tanke på att de är så markant vanligare respektive fåtaligare än alla övriga just bland de 59 valda arterna. Några djupare tolkningar kan inte göras utifrån Figur 1 eftersom arterna påverkar varandra ömsesidigt vad gäller position i rangskalan. Figuren visar att det är ungefär samma antal nedgraderingar som uppgraderingar.

I Figur 2 redovisas skattningarna i Ottosson m.fl. (2012) som procent av de korrigerade standardruttsvärdena. I Figur 3 sätts dessa procentvärden i relation till arternas abundans. De talrikaste arterna har skattningar som är ungefär desamma enligt båda metoderna. Däremot är värdena i Ottosson m.fl. (2012) klart lägre än de korrigerade standardruttsvärdena för de fåtaligare arterna. För arter med färre än 100.000 par är 90% av skattningarna lägre jämfört med 63% för arter med fler än 100.000 par.

Eftersom detta är det första försöket att använda korrigeringsfaktorer för att omvandla de relativa registreringarna längs standardrutterna till absoluta värden är det viktigt att fråga sig om de finska korrektionsfaktorerna är giltiga för ändamålet. Exempelvis ligger en stor del av Sverige söder om Finland, och Sverige har också mycket mera fjäll än Finland. Man har i Finland funnit att korrektionsfaktorerna varierar från norr till söder, bl.a. som följd av fågelsamhällenas totala tätheter, vilket skulle kunna innebära att omräkningsfaktorerna i södra Sverige kan vara annorlunda än i Finland. Det saknas dock helt studier för att bedöma detta. Däremot kan vi konstatera att de allmänna biotopmässiga förhållandena i Finland och Sverige i de flesta avseenden är ganska likartade, särskilt vad gäller de stora och dominerande arealerna av kommersiella skogar. Det finns därför a priori inga skäl att tro att de finska korrektionsfaktorerna skulle ge drastiskt missvisande resultat om man väljer lämpliga arter, dvs. arter där antalet observationer motsvara antal par.

Att de allra talrikaste arterna har likartade skattningar enligt Ottosson m.fl. (2012) och efter korrigering av standardrutternas linjetaxeringsdata är naturligt. För dessa arter fanns det gott om täthetsdata från nästan alla biotoper och alla delar av landet när beräkningarna gjordes. Det är också väntat att skillnaderna skulle bli större ju färre täthetsdata som fanns tillgängliga, dvs. för de fåtaligare arterna. Men det var inte väntat att skillnaderna mestadels skulle gå i en riktning. Jag kan inte säkert förklara varför skattningarna för fåtaliga arter i Ottosson m.fl. (2012) är lägre än de korrigerade räkningarna från standardrutterna. Eftersom jag själv var med och gjorde beräkningarna och skrev boken minns jag dock de diskussioner som vi hade. De mynnade ofta ut i att vi valde en konservativ och försiktig linje när vi saknade tillräcklig information; vi ville vara på den säkra sidan och inte överskatta bestånden. Jag tror att detta är en väsentlig orsak för flera av underskattningarna.

Eftersom det ännu inte finns några omräkningsfaktorer som gör att standardrutternas relativa täthetsvärden kan omräknas till absoluta tal är det viktigt att diskutera hur totalskattningar av de svenska fågelbestånden ska ske i framtiden. Det finns två sätt att göra detta.

Det ena sättet är att skapa ett nytt system av representativa provytor spridda i olika biotoper över hela landet för att på så sätt få täthetsdata som kan räknas upp till hela Sveriges areal. Eftersom antalet sådana provytor måste vara mycket stort och eftersom de måste karteras flera gånger under en häckningssäsong krävs stora arbetsinsatser och betydande kostnader. Jag bedömer därför att det i praktiken inte är troligt att ett sådant system kommer till stånd.

Det troligaste är, som jag redan framhållit, att standardrutterna inom Svensk fågeltaxering kommer att vara den huvudsakliga källan till kunskap om fågelbestånden, inte bara för tidstrender utan också för skattningar av absoluta beståndsstorlekar. Det andra sättet att skapa totalskattningar av bestånden är därför att experimentellt ta fram omräkningsfaktorer för standardruttsdata. Jag anser att detta bör göras genom att etablera ett mindre antal relativt stora provytor på lämpliga platser i landet. Dessa provytor inventeras sedan så väl att antalet häckande par blir känt med hög noggrannhet. I dessa provytor genomförs sedan punkt- och linjetaxeringar med samma metod som på standardrutterna och av ornitologer som representerar en genomsnittlig standardruttsinventerare. På det sättet får vi direkt omräkningsfaktorer som innefattar både lateral och basal korrigering och som är giltiga för svenska förhållanden. En variant är att förlägga sådana provytor i anslutning till befintliga standardrutter och utnyttja de data som redan insamlats under många tidigare år. En kombination av båda sätten är naturligtvis också möjlig.

Korta rapporter – *Short communications*

Mute Swan *Cygnus olor* raises Greylag *Anser anser* goslings

Knölsvan Cygnus olor *föder upp grågåsungar Anser anser*

HANS KÄLLANDER

On 24 May 2013, a pair of Mute Swans *Cygnus olor* was observed near the bird observation tower 'Almen' on the southern shore of Lake Krankesjön, South Sweden together with three newly hatched Greylag *Anser anser* goslings (Figure 1). The goslings were very closely associated with the swans picking small items from the surface that presumably were dislodged when the swans were foot paddling. Exactly what had happened before 24 May is uncertain but on 13 April the female swan had been incubating on a small islet close to the tower with a Greylag Goose incubating about a metre from her. On 29 May, i.e. five days later, the goslings kept closely to the female swan but one of the goslings was missing (parenthetically it can be noted that in recent years it has been more common than not that Mute Swans have, for unknown reasons, lost one or more young shortly after hatching). The 'family' was subsequently observed during much of the summer but the female Greylag Goose continued to incubate for quite a while, but it remains unknown whether any young left the nest.

On 18 June, the female swan with the two goslings was seen through a telescope from another bird observation tower on the eastern shore of the lake laying on the water some distance off the fringing reed beds near where they were first seen. No observations were then made until 22 July

Figure 1. The Mute Swan pair with three Graylag gooslings.
Knölsvanparet med tre grågåsungar.

when the female swan was feeding together with the goslings some distance out on the lake. When she swam back to the reeds to rest, the goslings followed her closely. Also five days later the 'family' was seen under similar circumstances and on 29 July the goslings were seen following the female swan both when she swam out on the lake to feed and when she returned to the reeds. After this, I checked the swans and the now quite large goslings on several occasions. On 10 August large flocks of Greylag Geese were resting on the water in the southwestern part of the lake. When the female swan, after a feeding bout, passed straight through the flock of geese, the now almost full-grown goslings followed her without showing any interest in their conspecifics. Two days later, however, the goslings appeared to have become a bit more independent than earlier and were seen further from the swan than seen previously and they swam back to the reeds alone while the female swan continued feeding. After this date, no more observations of the family were made.

Exactly how this adoption occurred is impossible to know but maybe the Greylag Goose had dumped a few eggs in the swan's nest. As the incubation period is shorter in the Greylag Goose than in the Mute Swan (28–29 vs 35–36 days; Bauer & Glutz 1968) and if the swan had laid at about the same time as the goose, one would expect the goose eggs to hatch about a week earlier than the expected hatching of the swan eggs. When the newly hatched goslings had started moving, this may have signalled to the swan that it was time to leave for the water. This, of course, is mere speculation. What is obvious, however, is that the goslings had become imprinted on the female swan.

What is most interesting in this case is that the goslings apparently grew and developed normally despite being forced to feed in an atypical environment, on the water rather than in a terrestrial habitat.

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Sammanfattning

Den 24 maj 2013 sågs ett knölsvanpar tillsammans med tre nykläckta grågåsungar i vasskanten utanför Almentornet vid Krankesjön (Figur 1). Gässlingrna var nära associerade med knölsvanarna och pickade material som virvlats upp när dessa ”fotpaddlade”. Knölsvanen hade tidigare ruvat på en liten holme utanför tornet med en grågås ruvande ungefär en meter från svanen. Den 29 maj, alltså 5 dagar senare, hölls gässlingarna nära svanhonan, men en av ungarna saknades. Grågåshonan fortsatte emellertid att ruva ett bra tag till men hade lämnat boet den 6 juni. Om några ungar kläcktes är okänt. Den 18 juni sågs knölsvanhonan med de två gässlingarna i tubkikare från Silvåkratornet, men sedan saknas anteckningar om dem till den 22 juli, då de födosökte tillsammans en god bit ut på sjön. När svanhonan sedan simmade in mot vassen för att vila, simmade gässlingarna tätt efter henne. Fem dagar senare sågs de igen under liknande omständigheter och den 29 juli följde gässlingarna svanen både då hon simmade in mot vassen och då hon senare ånyo simmade ut på sjön. Den 10 augusti låg stora svärmar av grågäss på sjöns sydvästra del. När svanhonan simmade in mot vassen med de nu nästan fullvuxna gässlingarna i släptåg, passerade de rakt igenom gåsflocken utan att gässlingarna visade något intresse för sina artfränder. Två dagar senare, dvs den 12 augusti, verkade gässlingarna dock vara något mer självständiga än tidigare och låg på vattnet en god bit från knölsvanhonan när hon födosökte och de simmade ensamma in mot vassen medan honan fortsatte födosöket ute på sjön. Efter detta datum gjordes inga fler observationer av knölsvan-grågåsfamiljen. Exakt hur denna adoption gått till är omöjligt att veta. Möjligt är att grågåsen dumpat några ägg i svanboet. Eftersom grågåsens ruvningstid är kortare än knölsvanens (28–29 dagar mot 35–36 dagar för knölsvanen (Bauer & Glutz 1968), och om svanäggen lagts vid ungefär samma tidpunkt som grågässens, skulle man förvänta sig att grågåsäggen kläcktes en vecka tidigare än den beräknade kläckningen för svanäggen. När gässlingarna sedan börjat röra på sig, kan detta ha signalerat till svanen att det var dags att uppsöka vattnet. Men detta är givetvis ren spekulation. Att gässlingrna präglats på knölsvanhonan var emellertid uppenbart.

Det intressanta i fallet är egentligen att gässlingarna uppenbarligen växte och utvecklades helt normalt trots att de tvangs födosöka i en för dem onormal miljö. Grågässen brukar föra sina ungar till fast mark, där de betar. Men i det aktuella fallet tycks gässlingarna ha funnit tillräckligt med föda trots att de hela tiden under sin uppväxt varit bundna till platser där knölsvanhonan furagerade, dvs ute på öppet vatten.

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A nest of European Golden Plover *Pluvialis apricaria* with all blue eggs lacking cryptic mottling

Ett bo av ljungpipare Pluvialis apricaria *med helt blå ägg utan kamouflageteckning*

SÖREN SVENSSON

While surveying birds within a study plot on alpine mores in southern Swedish Lapland, I found a nest of the Golden Plover *Pluvialis apricaria*. It contained four eggs that were all bright blue without cryptic colouration. The location was at about 65°50ʹN and 16°20ʹE, and the elevation about 800 meters. The nest was found at noon on 27 June 2014. The Golden Plover is common in the area and hundreds of nests have been found by our research group during more than fifty years. None of these has been reported to contain eggs without normal cryptic speckling. An incubating plover sits hard and when a nest is found it is most often because the bird is flushed or observed at close distance to sneak off the nest.

In the case reported here I discovered the nest at a distance of at least ten to fifteen meters because I got sight of the eggs in spite of having my attention primarily on counting birds in general. They were shiny blue or turquoise (Figure 1), very conspicuous and hence easily drew attention at some distance, which normal cryptic eggs never do.

Figure 1. Golden Plover nest with blue eggs without cryptic coloration that were easily detected at some distance. One of the parents lay dead and partly consumed by a predator near the nest. The eggs contained embryos that seemed to have developed normally until death.
Bo av ljungpipare med rent blå ägg utan skyddsteckning. De kunde lätt upptäckas på avstånd. En av föräldrarna låg död nära boet, delvis uppäten av någon predator. Äggen innehöll embryon som verkade ha utvecklats normalt tills de dog.

When approaching the nest, I found a dead adult about two meters from it. The eggs lay undisturbed in the nest cup with no sign of damage, indicating that the predator was most probably a bird, Gyrfalcon being the most likely guess. The eggs were wet because of rain and somewhat dirty, probably because of some fighting before the kill, indicated also by faeces on the eggs. It was a rather fresh kill, although the body as well as the eggs had cooled. The bird was probably the male as it is almost exclusively males that incubate during day-time (Whittingham et al. 2000), but as I did not examine the bird carefully it could have been the female as well. The eggs contained embryos that had been incubated about a week.

I have not approached any egg-collecting museums but one, the Swedish Museum of Natural History at Stockholm. In its collection of about 80 clutches there are no eggs similar to the ones I found (Ulf Johansson in litt.). I have not consulted the oological literature at large, only Rosenius (1937). He does not mention any aberrantly coloured eggs for the Golden Plover among 190 eggs. Whatever the exact proportion, non-cryptic eggs must be rare. And it seems that aberrant egg colouration of other kinds and in other species is also very rare, for example albinistic eggs which are only rarely found in otherwise normally coloured clutches (Gross 1968).

If a bird egg is coloured the pigment is either biliverdin or protoporphyrin or a combination of both (Kennedy & Vevers 1976). These two pigments are the only ones that have been found in any appreciable amount in bird eggshells. Consequently, these two pigments are responsible for the seemingly limitless variation of colours and patterns. Reflectance, concentration and a few other factors modify and enhance the variation created by the pigments alone. It is perhaps difficult to believe that two single pigments can be responsible for the enormous variation. However, in a recent study Hanley et al. (2015) demonstrated by a colour mixing model that this is actually possible. It has even been suggested that the same pigments may have governed the colouring of dinosaur eggs (Wienmann et al. 2015). From that it can perhaps be concluded that biliverdin and protoporphyrin were fixed as the both necessary and sufficient egg pigments at a very early stage of reptile–bird evolution. Interestingly, as mollusc shells are calcareous and have a great colourful variation – both these properties the same as in birds – it has been found that only protoporphyrin, not biliverdin, is responsible for the colour variation among molluscs (Verdes et al. 2015).

Biliverdin is the pigment that gives the blue to green colour (as in Starling) and protoporphyrin the brown to rusty colour (as in Peregine Falcon). This means that the shells of the Golden Plover eggs that I found had only the biliverdin pigment. There was no sign of the typical brown dots, patches or streaks of normal eggs. The black dots that are seen in the photographs were plant fragments. Regrettably, I did not collect any shell fragments. One might think that the absence of speckling in my case was caused by a defective outer shell layer, but this cannot be the case as protoporphyrin is mainly located in the deeper parts of the shell (Samiullah & Roberts 2013) and should have been visible if present at all. The content of the eggs including the embryos looked healthy so it is unlikely that any excess desiccation had occurred, indicating that the shell functioned as it should. This is in accordance with what Deeming (2011) found in his review of water loss in relation to egg colour; there was no effect of either the degree of speckling or the intensity of background colour.

It is well known that cryptic eggs increase nest survival among ground-nesting birds such as waders, for example in Western Snowy Plover (Colwell et al. 2011) and Moutain Plover (Skrade & Dinsmore 2013). Some cost is of course involved in laying eggs with cryptic colour as protoporphyrin and biliverdin must be synthesised in the shell gland for each egg, but it is likely that the increased reproductive success outweighs this cost (Wang et al. 2009). It is not possible to determine what may have happened in the present case, for example if absence of cryptic coloration of the eggs played any role for predation to occur.

Acknowledgment

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Sammanfattning

När jag inventerade fåglar på fjällhed i södra Lappland påträffade jag den 27 juni 2014 ett bo av ljungpipare som innehöll fyra helt blå ägg. Fyndet gjordes sydost om Ammarnäs på en nivå om 800 meter. I Ammarnäs har hundratals ljungpiparbon påträffats under de mer än femtio år som jag forskat i området. Normalt trycker ljungpiparna hårt på äggen och bon upptäcks därför vanligen när den ruvande fågeln stöts från boet eller observeras smyga iväg. Ett bo utan ruvande fågel upptäcks nästan aldrig spontant eftersom äggen är så väl skyddsfärgade. Det bo som jag rapporterar om här upptäcktes redan på tio till femton metes håll tack vare de lysande blå äggen. Närmare boet upptäckte jag en av föräldrarna ligga död ett par meter från boet. Fågel var delvis uppäten, gissningsvis dödad av en jaktfalk. Bofyndet illustreras i Figur 1.

Äggen saknade alla fläckar, prickar och streck som normala ägg har. De svarta prickar som syns på fotot var växtfragment. Boet var intakt med det fanns avföring på äggen. Detta tillsammans med växtfragmenten kan tyda på att viss strid förekommit på marken. Ljungpiparen hade dödats ganska nyligen men så väl fågeln som äggen hade svalnat. Äggen innehöll embryon.

Jag har inte gått igenom några äggsamlingar på större museer. På Naturhistoriska riksmuseet i Stockholm har Ulf Johansson gått igenom samlingen och meddelat att inga avvikande ägg finns bland 80 kullar. Inte heller har jag konsulterat den oologiska litteraturen i stort, bara Rosenius (1937), som inte nämner något avvikande bland 190 studerade ägg.

Om ett fågelägg är färgat så är färgämnet antingen biliverdin eller protoporfyrin eller en kombination av båda. Dessa två pigment är de enda som förekommer i nämnvärd mängd i äggskal. Det är alltså dessa som skapar den till synes oändliga variation som fågelägg uppvisar. Det kan vara svårt att tro att bara två pigment kan skapa denna variation. Men i en nyligen publicerad studie har Hanley m.fl. (2015) visa att detta är fullt möjligt. I en annan studie har Wienmann m.fl. (2015) kommit fram till att äggen hos dinosaurier också bör ha färgats av samma pigment. Från detta kan man kanske dra slutsatsen att de två pigmenten fixerades som både nödvändiga och tillräckliga på ett mycket tidigt stadium i utvecklingslinjen mot reptiler och fåglar. En intressant detalj i sammanhanget är att mollusker, som likt fåglarna har färggranna kalkskal, endast använder protoporfyrin.

Biliverdin är färgämnet som ger gröna eller blå skal (som stare) och protoporfyrin brun eller rostaktig färg (som pilgrimsfalk). Det betyder att de blå äggen i ljungpiparens bo endast hade biliverdin. Äggens innehåll var färskt och utan skador, liksom embryona. Det tyder på att skalen fungerat som de skulle under ruvningen trots att de saknade protopofyrin. Detta stämmer med vad Deeming (2011) fann i en översikt av vattenförluster, nämligen att uttorkningen inte påverkades vare sig av grundfärgens intensitet eller av mängden fläckar.

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