

High adult survival in a northern Eurasian Curlew (*Numenius arquata*) population

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Wetland birds such as waders are in decline across Europe. Information on demography is vital for assessing the causes of population declines, but estimates are needed from multiple populations and time periods. We used capture-recapture data from 2013–2022 including 134 individuals to estimate sex-specific apparent adult survival of Eurasian Curlews from a population breeding on agricultural fields in Finland, a stronghold region for this species in Europe. Using the Cormack–Jolly–Seber model that considers the recapture probability of individuals, we estimated apparent survival of adults to be 0.89 (SE 0.03) for males and 0.92 (SE 0.03) for females. Our estimates of adult survival are slightly higher than those previously estimated from Fennoscandian breeding grounds in 1980s–1990s (0.82–0.88). Thus, our results suggest that adult survival of populations breeding in northern Europe has not declined during the last decades. Our study supports the view that changes in reproductive success is the main cause of decline in the Eurasian Curlew populations.

1. Introduction

Waders have declined worldwide (Morrison *et al.* 1994, Stroud 2003, Amano *et al.* 2010, Simmons *et al.* 2015, von Numers *et al.* 2020). The causes of their population declines have been associated with changes in reproduction or survival using long-term data on demographic rates (Rönkä *et al.* 2006, Roodbergen *et al.* 2012, Piersma *et al.* 2016, Pakanen & Thorup 2016). Comparing demographic data from declining, stable and growing populations can help test the extent of and reasons behind the declines and plan conservation actions, which warrants research from

numerous population studies (e.g., Green 2002).

The curlews (*Numeniini*) include multiple declining and globally threatened species (Pearce-Higgins *et al.* 2017). They are large-bodied long-distance migrants characterised by low fecundity and high survival rates (Piersma & Baker 2000). Curlews are threatened both at their breeding and wintering sites (Pearce-Higgins *et al.* 2017). Habitat changes at wintering grounds are problematic as they may reduce survival and consequently cause dramatic declines in long-lived species (Piersma & Baker 2000). Therefore, information on demographic rates from all populations, including viable populations, are needed

to understand changes in population size and the factors threatening them (Pearce-Higgins *et al.* 2017, Viana *et al.* 2023).

In Europe, the Eurasian Curlew (*Numenius arquata*) is one of the meadow breeding birds that have declined strongly in recent decades (Roodbergen *et al.* 2012, Franks *et al.* 2017). The declines have been strongest in western and southern parts of their distribution in Europe (e.g., Franks *et al.* 2017, Brown 2015) with also recent range reductions in central and eastern Europe (Keller *et al.* 2020). The populations in northern Europe also show evidence of decline (e.g., Valkama *et al.* 2011, Lindström *et al.* 2019). The Finnish Eurasian Curlew population was recently categorised as near threatened, which means that the decline in population size does not exceed threshold rates of the threatened IUCN class (e.g., Lehtikoinen *et al.* 2019). Like most grassland breeding waders, their declines have been attributed to changes at the breeding sites, namely agricultural intensification, habitat changes and high nest and chick predation and their interactions (Grant *et al.* 1999, Valkama *et al.* 1999, Valkama & Currie 1999, Douglas *et al.* 2014, Franks *et al.* 2017).

Recent studies on adult survival measured from wintering populations suggest that survival is high and stable, and support the conclusion that the population declines are mainly linked to reproduction (Taylor & Dodd 2013, Méndez *et al.* 2018, Robinson *et al.* 2020, Cook *et al.* 2021). However, further information on survival is needed, including from regions supporting large remaining breeding populations, and also where older estimates are available, to enable comparison with recent survival estimates. Survival estimates from breeding Eurasian Curlews are from the 1980s–1990s (Ylimaunu *et al.* 1987, Berg 1994, Valkama & Currie 1999) but there is a lack of recent studies that have examined survival from breeding populations in Europe (Roodbergen *et al.* 2012, Viana *et al.* 2023).

We used capture-recapture data collected during 2013–2022 to estimate apparent adult survival of Eurasian Curlews from a population breeding on agricultural fields in northern Ostrobothnia, Finland and compared them to older survival estimates from northern breeding populations. On the basis of results from European

wintering grounds (Robinson *et al.* 2020, Cook *et al.* 2021), we hypothesized that adult survival of Eurasian Curlews breeding in Finland has not declined. As this region is a stronghold of breeding Eurasian Curlews in Europe (Valkama *et al.* 2011, Lindström *et al.* 2019, Keller *et al.* 2020), our results provide valuable information that can be used in assessing causes of population declines.

2. Material and methods

Our study population is located in agricultural areas in northern Ostrobothnia, Finland. This area is characterised by various types of fields (e.g., crop, fallow, hay) that have low vegetation and are suitable nesting and chick rearing environments for Eurasian Curlews. Curlew density in this region is among the highest in Europe (Keller *et al.* 2020). We collected data in counties Temmes, Tyrvävä, Muhos and Siikalatva (N64.179–N64.874, E25.507–E26.136). Starting from April, we scanned the breeding sites for Eurasian Curlews and territories multiple times per week and searched for their nests across the breeding season. We visited nests until known nest fate and ringed chicks. We caught incubating adult birds at the nest or when they were with chicks using walk-in traps that have one entrance hole. We replaced eggs with dummy eggs during the capturing of adults. We followed nests until hatching, and there was no evidence that catching causes abandonment. We did not systematically check for delays in resumption of incubation after trapping (Ewing *et al.* 2017), but the parents often changed incubation duties after capturing the other parent from the nest. When catching adults from broods, we waited for the parents to return to their chicks.

We sexed adults based on morphology (Summers *et al.* 2013) and marked them with a metal ring (Finnish ringing scheme) on the tarsus and placed a large plastic ring that included a large easily readable three-digit code formed by letters and numbers above the tarsus (Fig. 1). These codes were readable with a scope and returning birds with coded rings were photographed from a distance enabling their documentation. Verified readings of codes at the breeding sites were considered as resightings in the data



Fig. 1. A male Eurasian Curlew with the code ring (CNK) above the tarsus (photo taken by Reijo Kylmänen).

in the following years. Marking and resighting data constituted the capture–recapture data used to study survival.

Our sampling spread out in a large area (measured as polygon of ringing sites 794.1 km²) where effort varied greatly between field areas because the data were gathered entirely by Reijo Kylmänen. Uneven coverage of the study area can cause spatial variation in recapture rates (e.g., Cresspin *et al.* 2008). In addition, permanent emigration out of the study area may be more likely from areas situated closer to the edge of the study area or from areas that are covered less thoroughly (e.g., Pakanen *et al.* 2016). Both processes can violate the assumptions of the capture–recapture models by creating heterogeneity in recapture and survival probabilities (Lebreton *et al.* 1992). Controlling for such spatial variation in recapture

or apparent survival probabilities can be done by modelling them in relation a spatial parameter, for example distance to the center or edge of the study area when the study area is uniform (Marshall *et al.* 2004, Pakanen *et al.* 2016).

In order to control for possible spatial variation, we divided our study areas into two parts (variable “Area”, high effort areas vs. low effort areas). The “low effort areas” included large fields with a summed field area of 226.7 km² (Fig. 2). This large area reduced search effort per unit area meaning that these sites that were visited less often. The visiting rate in a given field in the low effort area was on average 3–4 times per week when the Curlews arrived in mid-April, but the rate declined as the spring advanced to about once per week by the end of May. The large area resulted in lower efficiency as there was less time to cover it but also because some areas were less accessible. We searched for Curlews mainly with scopes from the roadside which led to lower coverage in these large fields that included suitable habitat far away

from roads. This could potentially lead into the disappearance of individuals from our sampling, *i.e.*, permanent emigration. In certain years, some of the field roads in the low effort area remained inaccessible for vehicles, for example due to excessive flooding. We checked territories, especially those occupied in the previous year, more closely for Curlews and their nests by walking the fields. Marked Curlews were also searched for when they congregated to feeding groups in the evenings. The low effort area included 79 marked individuals.

The “high effort areas” included fields that were visited more often due to logistical reasons. Most importantly, the fields were small, and therefore search effort per area was high throughout the study (Fig. 3). The summed field area for the high effort area was 22.2 km². These field areas

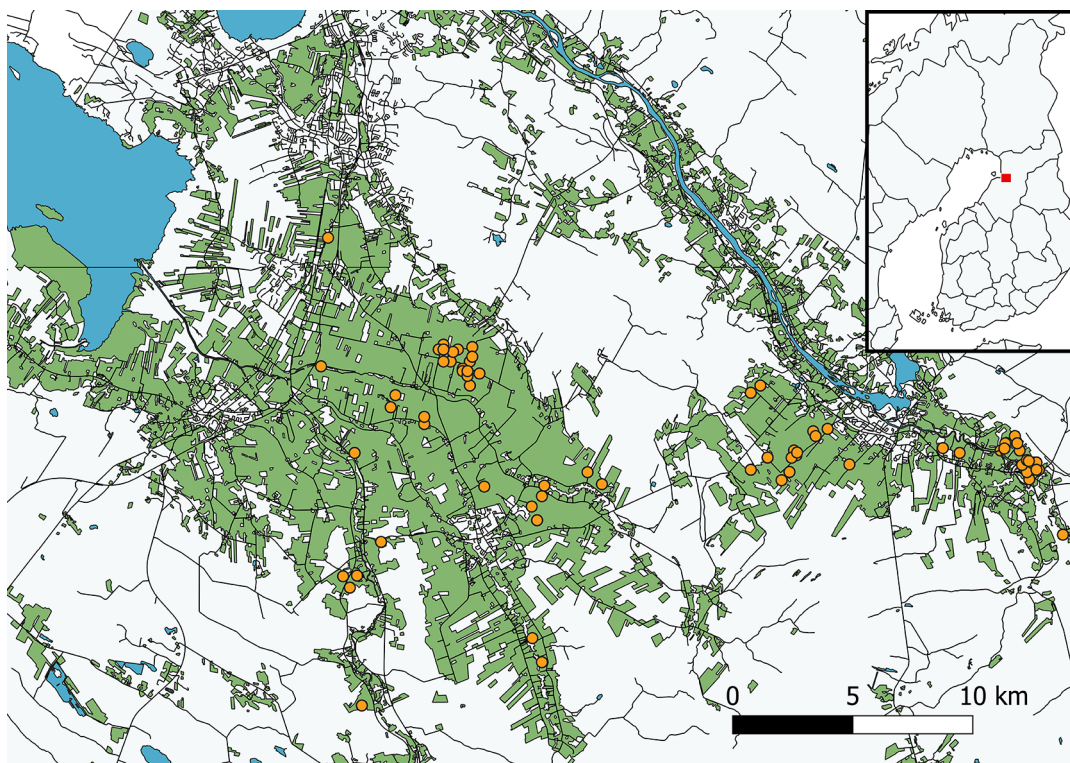


Fig. 2. Map showing the locations of the low effort study areas. Orange dots indicate ringing sites of individuals. Fields that are suitable breeding habitat of Eurasian Curlews area shown in green. Roads are black lines and water is shown in blue. National Land Survey open data, CC 4.0.

were easily accessible due to being near drivable roads which allowed coverage of whole fields and resighting of birds from inside a vehicle. These sites also stayed in the same condition across the study. These factors made visiting the sites easier and resulted in more visits and a more efficient resighting effort. When the Curlews arrived in the spring, our visiting rates per field were up to 2 times per day and the visiting rate continued to be high during the breeding season as these fields were under detailed study on nest success. We searched for territories and nests in all fields throughout the season by walking in the fields and observing parent behaviour from the vehicle or a long enough distance to corner down the nest location. Feeding groups were also searched for in the high effort areas during the evenings. These small field areas were more separated from other field areas (Fig. 3), which likely reduces the probability of permanent emigration to field areas outside of the study area. We thus expect

that recapture rates are higher in the high effort area and that the apparent survival estimates from the high effort area more reliably reflect true survival. The high effort area included 55 marked individuals. Altogether, the data included 134 individuals (males=77; females=57) and 386 observations from 2013 until 2022.

We analysed the data in program MARK using the Cormack–Jolly–Seber model (White & Burnham 1999, Lebreton *et al.* 1992). Because the data were too scarce and did not allow fitting of complex models, we did not examine temporal variation in survival or recapture rates. In addition to sex of the individual and the Area variable, we examined the effect of transience (permanent emigration after first capture) when estimating survival by using a time since marking variable (Tsm). This variable examines survival separately after first capture (residents+transients) and after subsequent resightings (residents; Pradel *et al.* 1997). Our starting model thus included

Sex, Area and Tsm and their interactions (*) for survival probabilities, and Sex together with Area for recapture probabilities $\Phi(\text{Sex} * \text{Area} * \text{Tsm})$ $p(\text{Sex} * \text{Area})$. This model fit the data (Bootstrapping GOF: $p=0.52$, $\hat{c}=1.00$).

We fitted a set of a priori models that were reduced version of the global model. We first modelled the recapture probabilities and used the best structure when modelling survival

probabilities. We used the Akaike's information criterion corrected for small sample size (AICc; Burnham & Anderson 2002) in model selection. We considered models with 2 AICc difference in model and variable support, and we used model averaging (models within 2 ΔAICc units) to control for model selection uncertainty when calculating survival estimates.

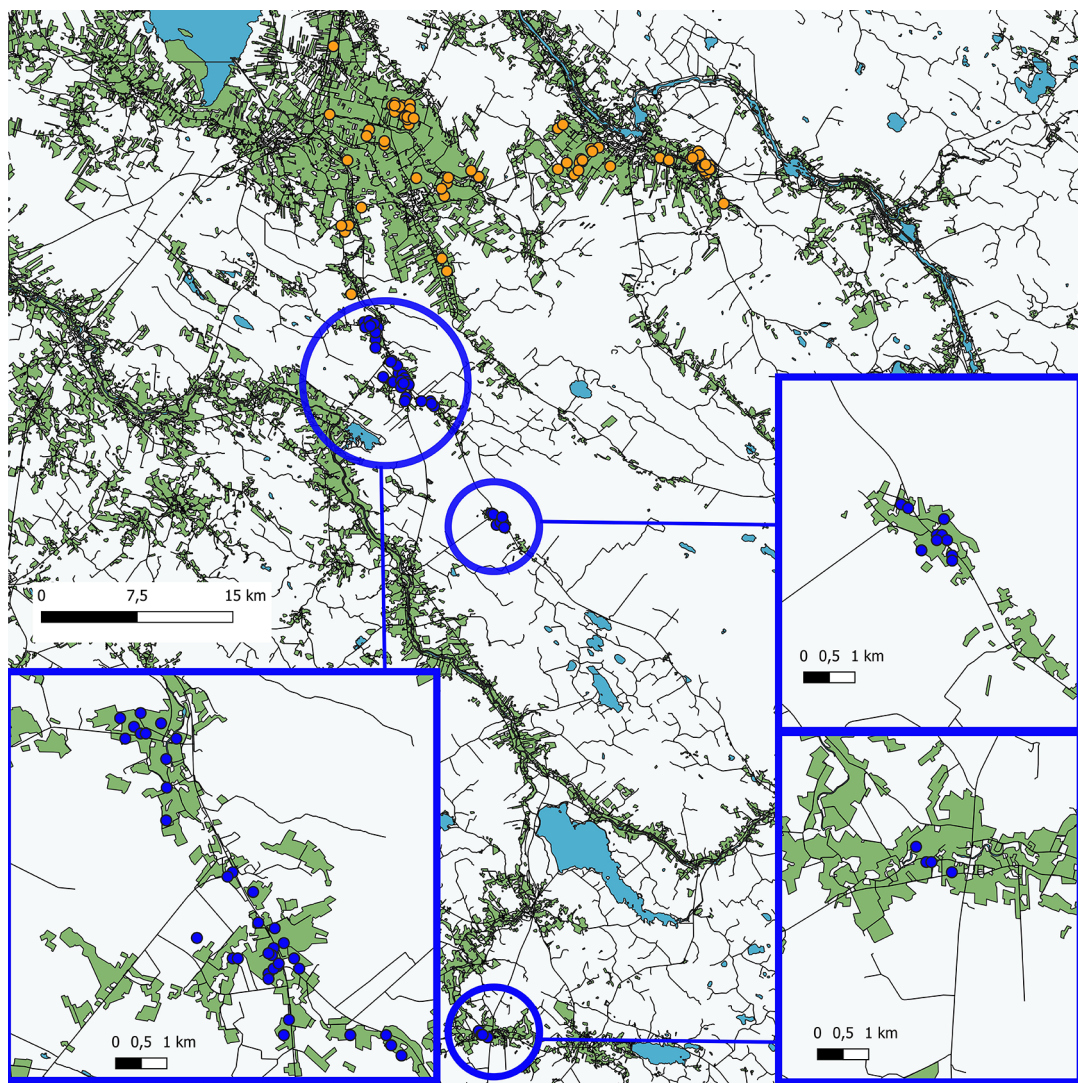


Fig. 3. Map showing the locations of the high effort study areas. Blue dots indicate ringing sites of individuals. Fields that are suitable breeding habitat of Eurasian Curlews area shown in green. Roads are black lines and water is shown in blue. National Land Survey open data, CC 4.0.

3. Results

The recapture probabilities were higher in males than in females (Table 1, $\Delta AICc=11.2$ model A1 vs. model A3) and lower in the low effort area (Table 1, $\Delta AICc=32.1$ model A1 vs. model A4; females 0.289, SE 0.046; males 0.486, SE 0.057)

than in the high effort area (females 0.658, SE 0.047; males 0.817, SE 0.033) being consistent with lower search and resighting effort and possible temporary emigration in the low effort area.
The best models ($\Delta AICc<2$) for apparent survival probabilities included effects of area,

Table 1. Modelling results for recapture probabilities (p) and survival probabilities (Φ) where: Area = low effort area vs. high effort area; Sex = sex of individual; Tsm = time since marking (two classes); Constant = no variation; + additive effects; * interaction. Models are ranked by the Akaike's information criterion corrected for small sample size (AICc). $\Delta AICc = iAICc - minAICc$; w = Akaike weight; k = number of parameters. Models with two or less interactions are written open in full.

Recapture probabilities					
#	Model	AICc	$\Delta AICc$	w	k
A1	$\Phi(\text{Area}*\text{Sex}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	783.59	0.00	0.728	11
A2	$\Phi(\text{Area}*\text{Sex}*\text{Tsm})\ p(\text{Sex}+\text{Area}+\text{Sex}*\text{Area})$	785.58	1.99	0.269	12
A3	$\Phi(\text{Area}*\text{Sex}*\text{Tsm})\ p(\text{Area})$	794.76	11.17	0.003	10
A4	$\Phi(\text{Area}*\text{Sex}*\text{Tsm})\ p(\text{Sex})$	815.65	32.06	0.000	10
A5	$\Phi(\text{Area}*\text{Sex}*\text{Tsm})\ p(\text{Constant})$	826.49	42.90	0.000	8
Survival probabilities					
#	Model	AICc	$\Delta AICc$	w	k
B1	$\Phi(\text{Area})\ p(\text{Sex}+\text{Area})$	778.77	0.000	0.160	5
B2	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	779.03	0.264	0.140	8
B3	$\Phi(\text{Sex}+\text{Area}+\text{Sex}*\text{Area})\ p(\text{Sex}+\text{Area})$	779.63	0.863	0.104	7
B4	$\Phi(\text{Sex}+\text{Area}+\text{Tsm})\ p(\text{Sex}+\text{Area})$	779.64	0.871	0.103	7
B5	$\Phi(\text{Area}+\text{Tsm}+\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	779.98	1.213	0.087	7
B6	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Sex}*\text{Tsm}+\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	780.31	1.544	0.074	9
B7	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Sex}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	780.53	1.762	0.066	8
B8	$\Phi(\text{Area}+\text{Tsm})\ p(\text{Sex}+\text{Area})$	780.82	2.054	0.057	6
B9	$\Phi(\text{Sex}+\text{Area})\ p(\text{Sex}+\text{Area})$	780.96	2.186	0.054	6
B10	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Sex}*\text{Area}+\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	781.14	2.369	0.049	9
B11	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Sex}*\text{Area})\ p(\text{Sex}+\text{Area})$	781.72	2.952	0.037	8
B12	$\Phi(\text{Sex}*\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})^a$	782.32	3.555	0.027	10
B13	$\Phi(\text{Sex}+\text{Area}+\text{Tsm}+\text{Sex}*\text{Area}+\text{Sex}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	782.63	3.861	0.023	9
B14	$\Phi(\text{Sex}*\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	783.59	4.820	0.014	11
B15	$\Phi(\text{Sex})\ p(\text{Sex}+\text{Area})$	787.23	8.462	0.002	5
B16	$\Phi(\text{Constant})\ p(\text{Sex}+\text{Area})$	788.46	9.686	0.001	4
B17	$\Phi(\text{Sex}+\text{Tsm})\ p(\text{Sex}+\text{Area})$	789.18	10.411	0.001	6
B18	$\Phi(\text{Sex}+\text{Tsm}+\text{Area}*\text{Tsm})\ p(\text{Sex}+\text{Area})$	790.33	11.562	0.000	7
B19	$\Phi(\text{Tsm})\ p(\text{Sex}+\text{Area})$	790.49	11.717	0.000	5

^a = no three-way interaction.

sex and time since marking. The model with the lowest AIC included only Area. Indeed, apparent survival was clearly higher in the high effort area than in the low effort area ($\Delta\text{AICc}=9.7$ model B1 vs. model B16; partial model coefficient on logit scale, β_{AREA} : from model B2: -0.969 , CI -1.533 , -0.405) suggesting that individuals may permanently emigrate more often from the low effort area. Survival of females tended to be higher than that of males but the models including sex were only 0.95 AICc units better than the reduced models (e.g., model B2 vs. model B5) and the confidence interval for the coefficient of sex overlapped zero (partial model coefficient on logit scale, β_{SEX} : from model B2 -0.505 , CI -1.109 , 0.080). The time since marking variable was also included the best models and two models included an interaction with Area. Although the confidence intervals for the coefficients overlapped zero ($\beta_{\text{AREA}*\text{TSM}}$: from model B2: -1.581 , CI -3.724 , 0.573). Model averaged estimates of apparent survival that consider model selection uncertainty (models B1–B7) indicated possible transience as survival of the first class of the time since marking variable (transients+residents) tended to be lower than survival in the second class of the time since marking variable (i.e., resident individuals) in the low effort area (Fig. 4). However, there was no evidence of transience

in the high effort area (Fig. 4.). Therefore, we retrieved survival estimates for the high effort area by model averaging models that did not include time since marking as factor. Survival of males was 0.891 (SE 0.032 , CI 0.811 – 0.939) and survival of females was 0.915 (SE 0.030 , CI 0.835 – 0.958) in the high effort area when time since marking was not considered.

4. Discussion

We estimated apparent adult survival of 0.89 – 0.92 for a Eurasian Curlew population breeding in agricultural fields in Finland. These estimates are in line with an estimate (0.90) derived from recovery data on chicks ringed in European breeding populations during 1968–2016 (Viana *et al.* 2023) and those estimated from wintering populations in Great Britain (0.91 – 0.92 ; Taylor & Dodd 2013, Robinson *et al.* 2020, Cook *et al.* 2021). Importantly, our estimate of survival for the last decade is slightly higher than those reported for breeding Eurasian Curlews in Fennoscandia during 1980s and 1990s (0.82 – 0.88 ; Ylimaunu *et al.* 1987, Berg 1994, Valkama & Currie 1999), thereby supporting the view that the survival of Eurasian Curlews breeding in northern Europe is high and has not declined during the last decades.

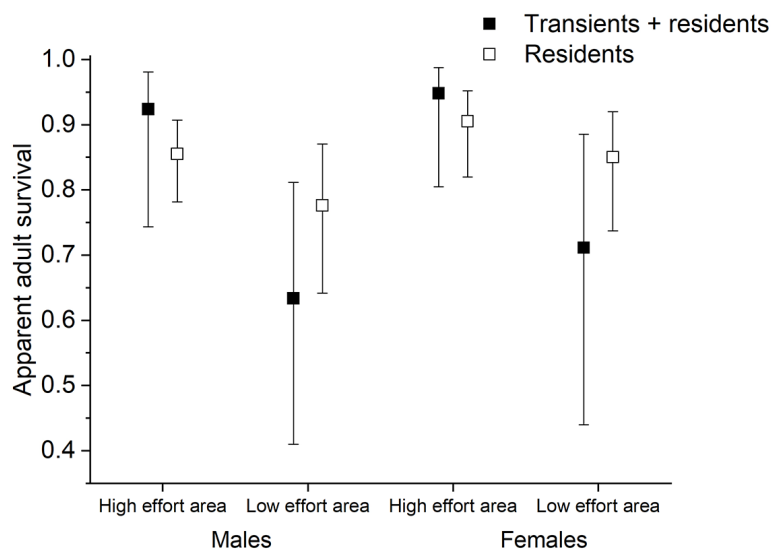


Fig. 4. Apparent survival estimates of male and female Eurasian Curlews separately for the low and high effort areas and for both time since marking classes (1: transient + residents; 2: residents only) derived by model averaging (models with $\Delta\text{AICc}<2$).

We controlled for recapture probabilities when estimating survival whereas the previous survival estimates for breeding Eurasian Curlews in northern Europe were return rates that may have been biased low. The lower survival in the earlier decades could have also resulted from hunting at the wintering grounds in Great Britain and France (Saurola *et al.* 2013, Robinson *et al.* 2020, Spina *et al.* 2022). Hunting was banned from 1982 onwards in Great Britain (Taylor & Dodd 2013) whereas in France a hunting ban was achieved only recently in 2020 with poaching remaining as a possible threat (Jiguet *et al.* 2021). Individuals wintering in France may have been subject to hunting during the 1980s and 1990s, but the number of Finnish-breeding birds reported shot has decreased dramatically in latest decades (Saurola *et al.* 2013, Pearce-Higgins *et al.* 2017). Curlews may also be susceptible to changes in their wintering areas, such as cockle dredging (Taylor & Dodd 2013), but also adverse changes in predation, weather and/or intensification of agricultural practices in wintering areas may reduce survival in particular years (Davidson 1981, Cresswell & Whitfield 1994, Franks *et al.* 2017, Cook *et al.* 2021).

Apparent survival estimates derived from open populations can be biased low by permanent emigration (Sandercock 2006), especially when suitable habitat extends beyond the study area (e.g., Pakanen *et al.* 2015). By using the Area-variable, we attempted to account for spatial heterogeneity in resighting probabilities and movement within the larger fields of the low effort area that likely resulted in permanent emigration. Indeed, both recapture and survival probabilities were higher in the high effort areas. Site fidelity of Eurasian Curlews to our high effort study area was clearly strong as their survival estimate was high. This is in line with reports on breeding site fidelity (Berg 1994, Valkama *et al.* 1998) and resightings/recoveries of breeding Finnish Eurasian Curlews (Saurola *et al.* 2013). At the low effort area, the large fields made it possible for individuals to emigrate out of reach but also other factors such as potential differences in nest predation, Curlew density or lower habitat quality may have affected dispersal (Berg 1994, Valkama *et al.* 1998). Furthermore, we found some evidence of transience (emigration after first capture) in

the low effort area but not in the high effort area supporting the view that survival probabilities in the low effort areas were biased low by permanent emigration.

We found that females tended to have higher apparent survival than males, which contradicts previous studies on Eurasian Curlews (Taylor & Dodd 2013). In general, females are the more dispersive sex in monogamous waders, including the Eurasian Curlew (Berg 1994, Kwon *et al.* 2022), and females should therefore be more likely to permanently emigrate from small study areas. The observed survival difference may, therefore, reflect for example sex-specific differences in predation pressure at the breeding site. Importantly, recapture probabilities were lower for females, which likely results from less conspicuous behaviour during the breeding season or possibly temporary emigration.

Adult survival estimated from this breeding population of the Eurasian Curlew breeding in Finland is high, being clearly within the expectation for a self-sustaining population (Brown 2015). This result agrees with earlier studies which suggest that the observed population declines in Europe are not linked to changes in adult survival but rather low reproductive success (Roodbergen *et al.* 2012, Robinson *et al.* 2020). Reproductive success is largely determined by nest survival, chick survival and local recruitment, which are mainly affected by conditions at the breeding sites among grassland breeding shorebirds (Plard *et al.* 2020, Pakanen *et al.* 2020). One of the main factors affecting success of Eurasian Curlews is predation of nests and young (Grant *et al.* 1999, Valkama & Currie 1999, Zielonka *et al.* 2019, Ewing *et al.* 2023). Increased predator numbers and new alien predators such as raccoon dogs (*Nyctereutes procyonoides*) threaten the ground nesting birds in Europe (Roos *et al.* 2018, Brzeziński *et al.* 2010, Holopainen *et al.* 2020, 2021). In our study area, the main nest predators include the raccoon dog, the red fox (*Vulpes vulpes*), corvids (*Corvus* sp.) and the marsh harrier (*Circus aeruginosus*) (own observations). Studies determining which mechanisms and habitat features at the breeding sites affect predation of nests and chicks should be helpful in designing management (e.g., Kentie *et al.* 2013, Laidlaw *et al.* 2017, Kaasiku *et al.* 2022, Pakanen *et al.* 2022, Ewing *et al.* 2023).

Suomessa pesivien isokuovien aikuissäilyvyys

Kahlaajapopulaatiot ovat vähentyneet Euroopassa laajalti. Demografinen tieto on hyvin tärkeässä osassa, kun arvioidaan populaatioiden vähenemisen syitä. Vähenemisen syiden selvittämiseen tarvitaan tietoa useista populaatioista ja eri aikajaksoilta. Isokuovi (*Numenius arquata*) on vähentynyt etenkin Länsi- ja Etelä-Euroopassa. Vähenemisen pääsyynä pidetään lisääntymismenestyksessä tapahtuneita muutoksia eikä aikuisten säilyvyydessä ole havaittu muutoksia talvehtimisalueilla tehdyissä tutkimuksissa. Pesivien populaatioiden aikuisäilyvyydestä on kuitenkin hyvin vähän viimeaikaista tietoa, jota voidaan verrata vanhempiin tutkimuksiin selvittääksemme mahdollisia ajallisia muutoksia, varsinkaan isokuovien pääpesimäpaikoilta Pohjois-Euroopasta. Me tutkimme Pohjois-Pohjanmaalla pesivän isokuovi-populaation aikuissäilyvyyttä lukurengasaineiston avulla vuosina 2013–2022 (134 rengastettua yksilöä). Analysoimme merkintätakaisinyntyaineiston Cormack–Jolly–Sebermallin avulla, jossa otettiin huomioon sukupuoli säilyvyydessä ja yksilöiden kiinnijäämistodennäköisyydessä. Aikuisten isokuovikoiraiden vuodenaikainen säilyvyys oli 0.89 ja naaraiden säilyvyys oli 0.92. Säilyvysestimaatit ovat hieman korkeampia kuin 1980- ja 1990-luvuilla havaitut säilyvykset (0.82–0.88). Nämä vanhempien tutkimusten tulokset voivat olla aliarvioita, koska niitä arvioitaessa ei huomioitu kiinnijäämistodennäköisyyttä. Tästä huolimatta näyttää siltä, että isokuovien aikuissäilyvyys ei ole vähentynyt viime vuosikymmeninä. Meidän tutkimustulostemme avulla voidaan arvioida isokuovien vähenemisen syitä Euroopassa. Ne tukevat aikaisempien tutkimusten tuloksia. Isokuovien väheneminen johtunee pääosin lisääntymiseen liittyvien vaiheiden (pesä- ja poikassäilyvyys) heikentymisestä.

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