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Department of Zoology, Institute of Ecology and Earth Sciences,
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I **Kaasiku, T.**, Praks, J., Jakobson, K., & Rannap, R. (2021). Radar remote sensing as a novel tool to assess the performance of an agri-environment scheme in coastal grasslands. *Basic and Applied Ecology*, 56, 464–475. https://doi.org/10.1016/j.baae.2021.07.002

II **Kaasiku, T.**, Rannap, R., & Kaart, T. (2019). Managing coastal grasslands for an endangered wader species can give positive results only when expanding the area of open landscape. *Journal for Nature Conservation*, 48, 12–19. https://doi.org/10.1016/j.jnc.2018.12.004

III **Kaasiku, T.**, Rannap, R., & Männil, P. (in press). Predation-mediated edge effects reduce survival of wader nests at a wet grassland-forest edge. *Animal Conservation*. http://doi.org/10.1111/acv.12774

IV **Kaasiku, T.**, Männil, P., Rannap, R. The effect of the golden jackal on nest survival of ground-breeding birds in a recently occupied part of the range. (Submitted manuscript)

V **Kaasiku, T.**, Carneiro, C., Rannap, R. Are artificial nests adequate to study wader nest survival and predation? A case on coastal grasslands. (Submitted manuscript)

Author's contribution to the studies (* denotes a moderate contribution, ** a high contribution, *** a leading role).

	I	II	III	IV	V
Original idea	***	*	***	***	***
Study design	***	**	***	***	***
Data collection	***	*	***	***	***
Data analysis	***	**	***	***	**
Manuscript preparation	***	***	***	***	**

1. INTRODUCTION

1.1. Open semi-natural wet grasslands and breeding waders

The current state of biodiversity and its future projections paint a bleak picture of the humankind's effect on the natural world. We have modified 75% of terrestrial land cover either directly or through indirect impacts (Brondízio et al., 2019). The leading process of land use change and, subsequently, loss of biodiversity, is driven by overexploitation and agriculture (Bellard et al., 2012; Maxwell et al., 2016). However, human actions through farming can also support biodiversity. For instance, low-intensity agricultural activities that imitate natural disturbances have created ecosystems for millennia (Pärtel et al., 2005). Such management of farmland, as opposed to intensive agriculture, still occurs today in specific, semi-natural habitats. These habitats are defined as "ecosystem with most of its processes and biodiversity intact, though altered by human activity in strength or abundance relative to the natural state" (IPBES, 2022). Semi-natural communities vary from grazed forests to periodically mown fens and several types of permanent grasslands all over the world.

Semi-natural grasslands are usually marginally productive, thus, they are threatened by the two extremes of agriculture, as their management is either intensified or ceased. Interestingly, both of these processes result in dense homogeneous vegetation that leads to the decimation of biodiversity dependent on open habitats (Busch et al., 2018; Sammul et al., 2012). All over Europe the socio-economic changes during the last century have strongly affected the management of semi-natural habitats – a system that used to be relatively stable for hundreds of years (Hooftman & Bullock, 2012; Luoto et al., 2003). The remaining area of semi-natural grasslands in Northern Europe is now estimated to be only a small fraction of what it was in the beginning of the 20th century (Eriksson et al., 2002; Herzon et al., 2021; Luoto et al., 2003).

Waders, a polyphyletic group among *Charadriiformes*, are mostly migratory ground-nesting birds dependent on multiple types of open landscapes both during the breeding and non-breeding season. Primarily as a result of habitat loss, they are threatened both in Europe and globally (Pearce-Higgins et al., 2017; Thorup, 2006). Wet grassland breeding waders is an umbrella term for the species mostly inhabiting, as the name suggests, various types of wet grasslands during the breeding season. These species have also recently gone through a substantial population decline as a whole in the European Union (EU) due to low breeding success (Leyrer et al., 2018). While the indirect reasons behind low reproduction rates are complex, there is a consensus about their anthropogenic origin: land use change, principally through wetland conversion to arable land and drainage; but also intensive agriculture; land abandonment; climate change; facilitation of the spread and abundance of nest predators (Leyrer et al., 2018).

This thesis focuses on a habitat at the border of semi-natural grasslands and wetlands – the Boreal Baltic coastal grassland. Although a specific habitat type, the topics relevant to it are inherent to other managed open wetlands, such as salt marshes and meadows, grazing marshes, brackish swamps, and freshwater marshes, but also other permanent or improved grasslands that host breeding wader populations. The Boreal Baltic coastal grassland is an early successional habitat listed in the Annex I of the EU's Habitat Directive (Natura habitat code 1630* – Boreal Baltic coastal meadow) (European Commission, 2013). These grasslands are shaped by the effect of postglacial land uplift, floods, sea ice, and grazing by livestock and wild geese (Auffret & Cousins, 2018; Jutila, 2001). Semi-natural grasslands represent a wide variety of services, including the well-being of human society though supporting rural areas and enabling low-intensity farming (Herzog & Schüepp, 2013; Hooftman & Bullock, 2012; Villoslada et al., 2018). Moreover, coastal grasslands are essential to biodiversity and their management has been shown to support several species groups (Rannap et al., 2017). For example, in Estonia, this unique habitat hosts ca. 30 protected plant species (Kose, 2021); amphibians, including the threatened natterjack toad (*Epidalea calamita*) (Rannap et al., 2007); and accommodates several protected birds species (Rannap et al., 2017), either as a breeding habitat or a stopover site on migration. Similarly to other types of semi-natural grasslands, ca 90% of coastal grassland habitat was lost in Estonia during the last century, mostly as a result of land abandonment (Luhamaa et al., 2001). Furthermore, an active afforestation programme of low production open habitats took place in 1960s (Palang, 1998) (Fig. 1). The change in bird communities that follows land abandonment in Boreal Baltic coastal grassland habitat has been eloquently described by Soikkeli & Salo (1979) in Finland, where open habitat specialists were quickly replaced by reedbed-favouring species.

Figure 1. This 55-year-old Scots pine (*Pinus sylvestris*) plantation is a typical woodland bordering a semi-natural coastal grassland in Estonia. Paope, Hiiumaa, April 23, 2020.

For Boreal Baltic coastal grasslands, waders have been suggested to be the best indicators of habitat condition, as they disappear from areas with high, overgrown vegetation (Puurmann & Kokovkin, 2005; Soikkeli & Salo, 1979). In Estonia, these grasslands provide habitat for dense wader communities, including globally non-threatened species with decreasing population trends such as the common ringed plover (*Charadrius hiaticula*) and the common snipe (*Gallinago gallinago*), but they also support near-threatened species such as the northern lapwing (*Vanellus vanellus*), the black-tailed godwit (*Limosa limosa*), the Eurasian oyster-catcher (*Haematopus ostralegus*), the Eurasian curlew (*Numenius arquata*), and the great snipe (*Gallinago media*) (IUCN 2021). Importantly, coastal grasslands also sustain the southern dunlin (*Calidris alpina schinzii*) and the ruff (*Calidris pugnax*) which are focus species for this habitat management in Estonia (Fig. 2). The breeding populations of these two species, that were still common in the middle of the 20th century, have crashed across the region (Birdlife International, 2021; Herrmann & Thorup, 2009). Nevertheless, Estonia still hosts half of the southern dunlin's Baltic Sea population (Herrmann & Thorup, 2009). Based on the monitoring scheme of coastal grassland breeding birds, the northern lapwing and the common snipe show contrasting trends to the European-wide declines (Environment Agency, 2019; Leyrer et al., 2018). It can be speculated that this could be resulting from the habitat restoration effort carried out over the past two decades. However, all other species are in moderate (1–5%) or steep (>5% annually) decline (Environment Agency, 2019). As these declines are occurring simultaneously with habitat restoration and management, it raises the concern, whether the currently fragmented landscape still holds suitable conditions for breeding waders.

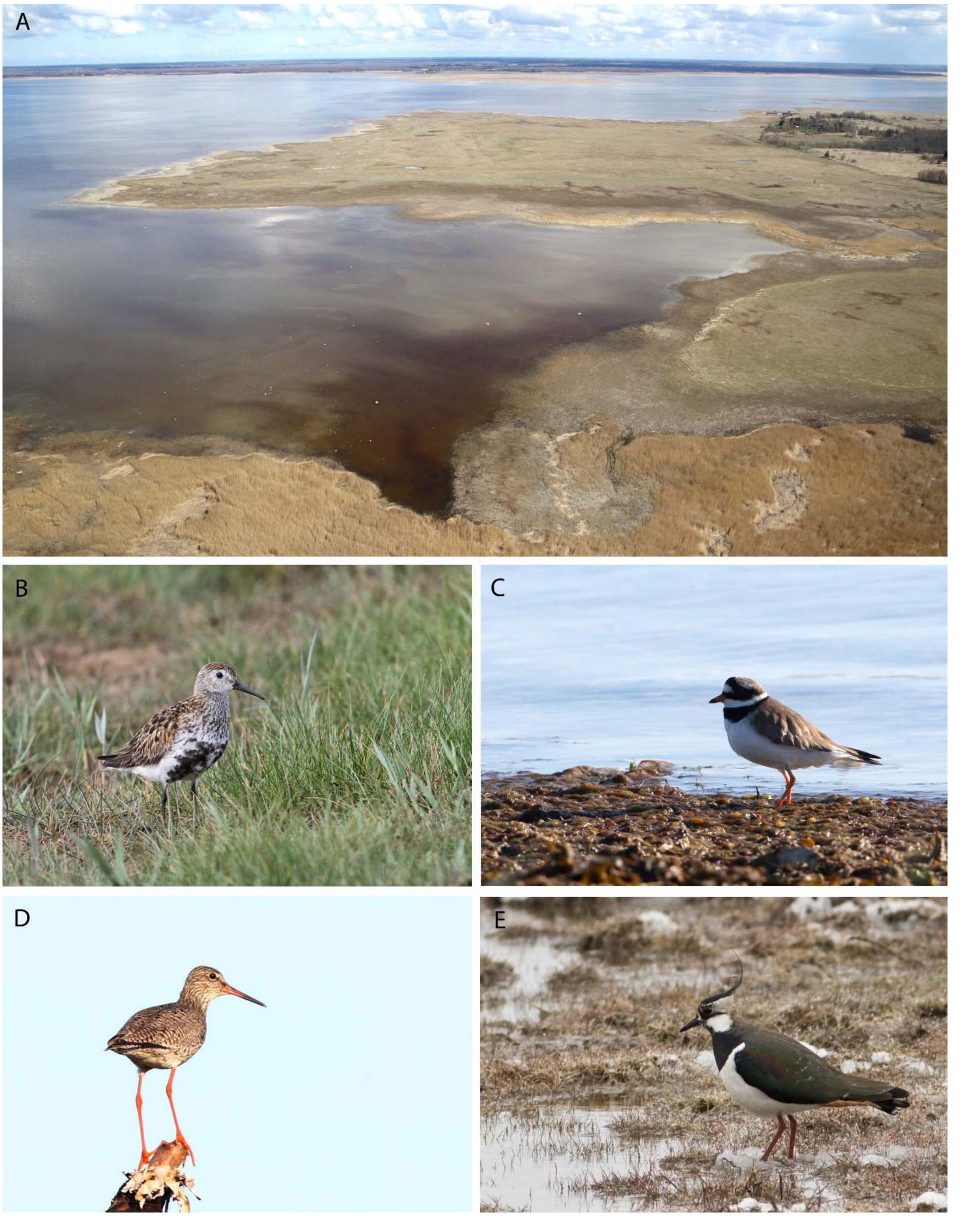

Figure 2. (A) A fragment of the 300 ha Salmi coastal grassland in Matsalu NP, Estonia (photo from May 10, 2017). A shallow bay surrounded by winding coastline; higher and lower parts with less or more waterlogged soils; areas of short grass mixed with patches of reedbed in soft and wet fields where cattle doesn't graze. This site is the core breeding area for the southern dunlin (B) in the Baltic Sea region, other confirmed breeding wader species include the ruff, the black-tailed godwit, the Eurasian curlew, the common ringed plover (C), the common redshank (*Tringa totanus*) (D), and the northern lapwing (E).

1.2. The state of Boreal Baltic coastal grasslands as breeding ground for waders

After decades of land abandonment, a positive change for the coastal grassland habitat took place in the early 1990s with the reform of the nature conservation policy, when the idea of supportive interaction between humans and nature began to take root in Estonia (Puurmann & Kokovkin, 2005). Since then, habitat restoration projects and subsequent continuous grazing activities on restored sites have been carried out in close cooperation with landowners and managers. Although habitat restoration has proven to be time-consuming, the area of abandoned coastal grasslands has decreased over the last 20 years in Estonia (Kose et al., 2020). Currently, subsidies from the EU's agri-environment scheme (AES) are applied on ca 11 500 ha of coastal grasslands to support biodiversity-friendly agriculture (Agricultural Registers and Information Board, 2020).

Agri-environment subsidies are the main mechanism of supporting biodiversity in the vast agricultural land in the EU. While having a high potential impact, their effectiveness in benefiting biodiversity is sometimes questioned (Kleijn et al., 2006; Kleijn & Sutherland, 2003). Although the area under management on Estonian coastal grasslands has increased due to the AES subsidy, it has not been investigated whether the management scheme has been effective in ensuring higher habitat availability for breeding waders. Since the AES measures differ in their local requirements and are therefore not directly comparable, similar schemes elsewhere have been shown to support targeted wader species or meadow-bird communities (e.g., Franks et al. 2018; Kleijn et al., 2006; Schekkerman et al., 2008; Smart et al., 2013), although not always working as efficiently as expected (Kleijn et al., 2001; Mason et al., 2019; Verhulst et al., 2007). Knowing the effectiveness of the scheme will make it possible to remedy its potential shortcomings and increase well-managed areas for biodiversity (Kleijn et al., 2001). Nevertheless, studying only habitat availability does not provide the full picture about key habitat features vital for breeding waders.

Understanding how species interact with their environment is essential for discerning the issues of habitat loss. Wet grassland breeding waders spend a varying, but relatively short time of their annual cycle in the breeding grounds. They arrive from migration, restore their energy reserves, mate, and start nesting. Thus, the habitat needs to provide enough resources for the pre-laying period, incubation, and chick rearing time. Grassland-breeding waders prefer open landscapes, which probably ensure better predator detection ability (Amat & Masero, 2004; Koivula & Rönkä, 1998) and allow them to perform elaborate aerial displays (Gunnarsson, 2020). Their nest is a simple scrape on the ground, either open or concealed by vegetation, and their chicks are precocial, leaving the nest cup soon after hatching. As chicks need to move and feed independently from the beginning of their lives, while being under parental protection, the nesting site needs to be in accordance with specific requirements: foraging areas must be close, accessible for small chicks, and ensure an adequate supply of food (Fig. 3).

Wader chicks mostly feed on arthropods, and since many arthropods depend on waterbodies for at least some period of their life cycle, wet places are essential as foraging areas (Buchanan et al., 2006; Eglington et al., 2010; Johansson & Blomqvist, 2010). Many wader species are extremely site-faithful (Jackson, 1994), which in the case of habitat conditions, means that the same location must meet all the requirements during each breeding season. The Boreal Baltic coastal grassland context can be used as a model to investigate habitat selection of waders in formerly abandoned and overgrown, now increasingly restored, yet fragmented, areas. This allows the following problems to be investigated: Do waders avoid breeding near reedbeds or close to patches of forest that fragment the landscape? Do they choose their territory close to freshwater pools or the brackish shoreline? Does the ground height influence their habitat selection on these low-lying pastures? Knowing the answers to these questions will help with designing adequate habitat restoration and management plans.

Figure 3. Southern dunlin chick in a probably good "chick habitat" – sparse vegetation makes it easy to walk around, grass is high enough to cover the chick from possible predators and wet conditions facilitate abundance of food. Photo from Õunaku coastal grassland, Hiiumaa, May 19, 2018.

While habitat restoration and management are positive steps towards increased connectivity of remaining coastal grasslands (Kose et al., 2020), the occurred habitat loss has inevitably caused habitat fragmentation (Fig. 4). Effects associated with fragmentation include reduction of the habitat patch size, increased isolation from other suitable habitat and increased edge effects (Fahrig, 2003). The latter is a much-studied topic: while first, since describing the phenomenon in the 1930s (Yahner, 1988), the increase of habitat edge through fragmentation was seen as a way to support biodiversity, a wave of studies in 1980s and onwards have illustrated the adverse effect of an edge. One such example is the hypothesis that avian nest survival increases with the distance to habitat edge (Batáry & Báldi, 2004; Chalfoun et al., 2002; Stephens et al., 2004). While this theory has found support in most studies carried out in forests (Andrén, 1995; Batáry & Báldi, 2004), research focusing on open landscapes has often failed to prove this pattern (e.g., Bertholdt et al., 2017; Ottvall et al., 2005; Valkama & Currie, 1999; but see: Ellis, 2020; Johnson & Stanley, 1990; Møller, 1989). However, as demonstrated by studies showing lower wader densities in more fragmented landscapes (Amar et al., 2011; Douglas et al., 2014), the topic of edge effects is potentially of high importance for breeding waders.

Figure 4. Today, a typical coastal grassland habitat is fragmented by patches of forest. Such fragmented habitat can potentially support higher numbers of generalist predators. Photo of Taguküla islet, Hiiumaa, May 10, 2017.

The direct cause behind the edge effect on nest survival is predation (Chalfoun et al., 2002). During the last decades, the numbers of mammalian mesopredators have increased both in Estonia and Europe as a whole (McMahon et al., 2020; Veeroja et al., 2021). The mesopredator release theory proposes that the rise of mesopredators can be attributed to the extirpation of apex predators from ecosystems (Prugh et al., 2009; but see Castle et al., 2021), while their numbers have also been suggested to grow as a response to resource subsidies from humans, among other factors (Elmhagen et al., 2017). While habitat loss has undoubtedly been the driving force behind wader population declines, it has been shown that high nest predation rates of ground-breeding birds are contributing to these declines and to the unsuccessful recovery of wader populations (McMahon et al., 2020; Roos et al., 2018). Furthermore, it has also been demonstrated that degraded habitats can support increased predator densities (Evans, 2004; Klug et al., 2009). Unravelling the relationship between habitat loss and predators has nevertheless remained a knowledge gap, hampering the protection of waders (Laidlaw et al., 2021).

In addition to habitat, predators can also be affected by interactions within the predator guild. Over the last decade the predator guild of Estonian coastal grasslands has been supplemented with a novel predator for the region, the golden jackal (*Canis aureus*). This widespread carnivore species that ranges from Central Europe to Southern Asia has recently gone through an explosive range expansion (Hoffmann et al., 2018; Spassov & Acosta-Pankov, 2019), shifting its range edge 1500 km north- and westwards during the last decade. Reaching new areas, it ignites old concerns: will it prey on livestock, transmit pathogens and/or affect local prey species through its impact on trophic dynamics (Maran, 2015; Rutkowski et al., 2015)? Although the golden jackal might have been present in Estonia a few years earlier, the first confirmed record of the species dates to 2013 (Männil & Ranc, 2022). In the same year, local environmental officials interviewed for their reactions towards this novel species expected it to become a competitor to other mesopredators, such as the abundant red fox (*Vulpes vulpes*) (Maran, 2015). From the one hand, there is indeed some evidence that the golden jackal could restrain the local red fox populations (Scheinin et al., 2006), therefore reducing the pressure on prey items. On the other hand, as a generalist predator it is likely to be able to influence local prey populations (Stoyanov, 2012). The golden jackal established its core population in Estonia in the vicinity of coastal grasslands, creating a unique natural setup for studying its effect on the nest survival of endangered wader species and interactions with other mesopredators. Obtaining knowledge on the effect of the golden jackal on it prey is essential to avoid predator management decision-making based on fear or assumptions.

Studies on nest survival or nest predation patterns rely on the availability and ability to discover the nests. However, searching for bird nests is time-consuming, as they are often difficult to find, and it increases the disturbance to breeding birds. Thereby, an alternative method that uses artificial nests instead of natural bird nests has been used (e.g., Møller, 1989; Olsen & Schmidt, 2004). In this approach, artificial clutches that mimic real bird nests are laid out to study nest survival patterns. Although this method gives the opportunity to test different experimental designs to study nest predation, this practice has been questioned by multiple authors (Berg, 1996; Major & Kendal, 1996; Pärt & Wretenberg, 2002), as it has been shown that artificial nests can differ in their predation rates and/or predator species (Bentzen et al., 2017; Davison & Bollinger, 2000; Willebrand, 1988). To be able to diminish the probable bias in the results of potential future studies with artificial nests, it has been suggested that their use should be validated locally.

1.3. Objectives of the thesis

The main objectives of this thesis are to assess the availability of coastal grassland habitat as a nesting site for waders, to understand waders' habitat selection and factors that influence it, and to provide practical nature conservation information to improve the state of wader populations breeding on coastal grasslands. Specifically:

(I) Using remote sensing data, I aim, for the first time, to assess the efficacy of the agri-environment scheme applied on coastal grasslands in creating suitable habitat for waders in terms of vegetation height. Potentially, this also allows to estimate the area of the remaining suitable coastal grassland habitat for waders at a country-wide scale.

(II) While coastal grassland habitat is shaped by several abiotic and biotic factors, its ultimate state is mostly dependent on agricultural management. I aim to reveal the habitat characteristics that limit or promote territory establishment of breeding waders in the system of recently restored but nevertheless fragmented grasslands.

(III) Habitat loss leads to habitat fragmentation and, consequently, to edge effects. I aim to understand whether adverse edge effects on nest survival could contribute to wader population declines in the Boreal Baltic coastal grassland system, where wet grasslands are fragmented by patches of forest.

(IV) A novel predator species for the region, the golden jackal, has recently gone through a large-scale range expansion. I aim to unravel its impact on nests of ground breeding birds and its interaction with other mesopredators species.

(V) For measuring nest survival, artificial nests have previously been used, and could possibly be used also in the future, as a cost-effective alternative to conduct natural nest survival studies. I aim to verify the reliability of artificial nests as a proxy for natural wader nests in the coastal grassland habitat.

2. METHODS

2.1. Study design and data collection

All study areas are located in the western part of Estonia, where most of the coastal grassland habitat occurs (Fig. 5). These coastal grasslands are situated in the hemi-boreal climate zone, between latitudes 58–59º N, characterized by cold winters (mean temperature in December-February –2.6 ºC), warm summers (mean temperature in June-August 16.4 ºC), and moderate precipitation (mean 662 mm/year; Estonian Weather Service, 2021).

Study **I** covers 21 900 ha of Boreal Baltic coastal grasslands, which form approximately 90% of the total area of this habitat type currently present in Estonia (Helm & Toussaint, 2020). The ground truth data for the study was collected on ten coastal grassland sites. Subsequently, the same ten sites were used in studies **III** and **IV** and seven of these sites were also selected for study **V**. Study **II** was conducted on 23 coastal grasslands, seven of which overlapped with other survey areas (Fig. 5). Each study site formed a coherent management unit that had been used as a pasture for at least five consecutive years by the start of the study (Agricultural Registers and Information Board, 2020).

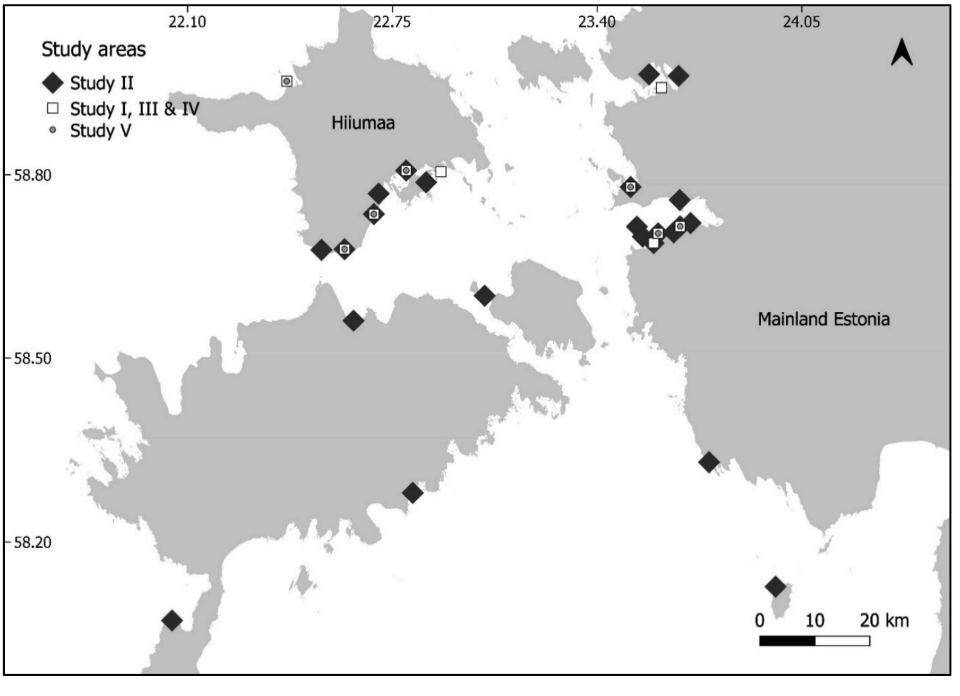

Figure 5. Map of study areas in western Estonia. Several sites were included in multiple studies, hence the icons denoting different studies overlap.

Studies **I** and **V** were carried out in a single year, in 2019 and 2020, respectively. Study **II** covered the breeding seasons of 2012 and 2013, while studies **III** and **IV** spanned over three breeding seasons, from 2018 until 2020. Studies **I, III**, and **IV** concentrated on the whole breeding wader community, study **II** included four of the most abundant species (southern dunlin, common snipe, common redshank, and northern lapwing) while study **V** included information on typical open-nesting species (the northern lapwing and the common ringed plover).

Study **I** is based on the assumption that high-growing plants indicate the effectiveness and quality of habitat management, as a species-rich coastal grassland plant community mostly consists of low-growing species (Kose, 2021). The vegetation of Estonian coastal grassland habitat was classified using remote sensing (European Space Agency's Sentinel 1 satellite (Bourbigot et al., 2016)) and ground-truth data collected in 345 locations. Ground truth data was collected during the wader breeding season, from April 17 until June 23 in 2019. Since it has been shown that synthetic aperture radar is not able to distinguish between small differences in vegetation height (Cimbelli & Vitale, 2017; Voormansik et al., 2013), vegetation was measured in two categories: <50 cm=*suitable*, >50 cm=*unsuitable* for breeding waders (Fig. 6).

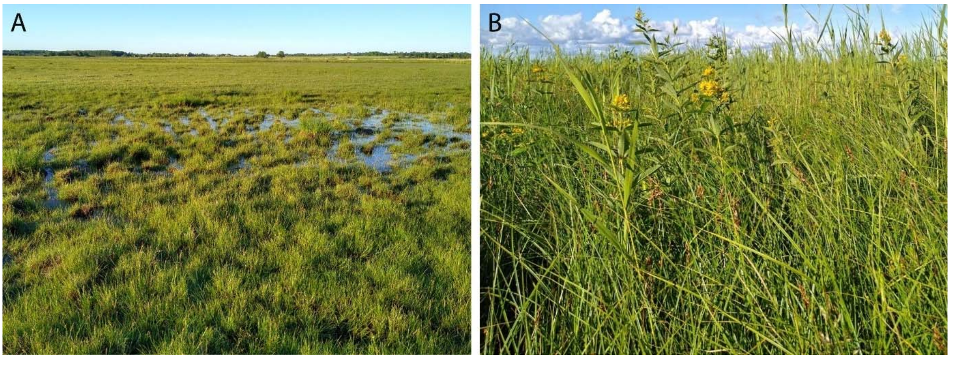

Figure 6. *Suitable* (A) and *unsuitable* (B) habitat for breeding waders in terms of vegetation height as classified in study **I**.

For each reference data point, radar image pixel values were obtained from the image with the nearest date, using 13 synthetic aperture radar images in total (acquired between April 15 and June 21, 2019). To be able to evaluate the effect of the AES, different semi-natural community (SNC – the name of the local management scheme) map masks derived from the database of Estonian Agricultural Registers were used. Satellite images were pre-processed and associated to the two vegetation groups based on ground-truth measurements. To support the habitat classification with wader nest location data, nests were searched for on coastal grasslands. More detailed information about the methods used can be found in the Materials and Methods section of study **I**.

The aim of study **II** was to assess habitat characteristics that influence waders' habitat selection. Locations of breeding wader territories were mapped in early mornings from the 10th of May until the end of May. Concurrently, habitat characteristics (reed beds, wet areas, unmanaged areas, and ditches) which were previously considered as possibly important for waders to establish breeding territories, were mapped in the field. The open coastline and forest were outlined from aerial images (Estonian Land Board, 2013). Additionally, the height of ground level was derived from 2013 LiDAR data (Estonian Land Board). Exhaustive description of the methods is given in the Materials and Methods section of study **II.**

For studies **III** and **IV** nest survival of the breeding wader community was recorded on ten Boreal Baltic coastal grassland sites in Estonia. Study sites included some of the most important and largest coastal grasslands, as well as isolated and smaller areas. The surveyed area per site ranged from 19 to 91 ha (mean=41 ha, total=408 ha). Nests of all breeding wader species were searched for. Upon finding a nest, the species was identified, two eggs were floated to determine the incubation stage following Liebezeit et al. (2007), and nest location was recorded. Nests were searched for from the beginning of the breeding season until no known nests were present on sites. Nests were visited after an average of 8 days (range 5–11) to check whether the nest was still active or lost (to predation, abandonment, flooding) (Fig. 7). In 2019 and 2020 camera traps were deployed on a subset of nests (n=85), to record predation events and predator detection rates. Thorough details on the methods, including the determination of nest fate, can be found in the Material and Methods sections of studies **III** and **IV**.

Figure 7. (A) A common redshank nest cup with a depredated egg next to it. (B) A northern lapwing nest with a recently hatched chick and two eggs

To be able to assess the effect of habitat features on nest survival in study **III**, trees and forest in the proximity of study sites were mapped from aerial images taken in May 2018 (Estonian Land Board). A patch of trees or bushes was classified as "tree" if the width of the patch was less than 30 m, all woodland patches wider than that were classified as "forest". Distances between each nest and the closest tree and forest edge were measured. To identify the effect of forest cover at a landscape scale, a buffer of 1 km around each nest was created and the percentage of the area under forest cover inside this buffer was calculated. For this, the extent of forest cover obtained from the base map of the Estonian Land Board (2020) was used.

For study **V** an artificial nest experiment was carried out to comparatively assess the survival of nests of common open-nesters (northern lapwing and common ringed plover) and artificial nests. After finding a natural nest, a clutch of 4 domestic quail (*Coturnix* sp) eggs was placed ca 100 m from it in a shallow nest cup scraped with the heel of a boot (Fig. 8). Additionally, to determine nest predators for the two types of nests, cameras were placed on both artificial and natural nests. Nests were revisited after an average of 8 days to record their status. Artificial nests were considered as "hatched" at 27 days of age.

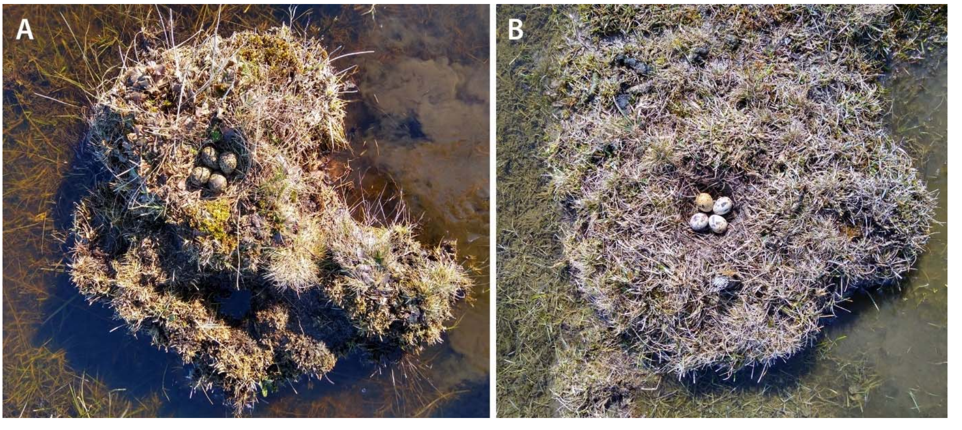

Figure 8. Comparison between the northern lapwing (A) and an artificial (B) nest.

2.2. Statistical analyses

As a first step in study **I**, Sentinel-1 images were pre-processed. The reference data were then divided into training (230 values) and test (115 values) sets. Corresponding backscatter values from single pixels were retrieved for ground truth data from pre-processed radar images and a t-test was performed on the training set to determine whether backscatter differed between habitat classes. The classification threshold for distinguishing *suitable* and *unsuitable* areas in the training set was based on the ROC curve. As a next step, the acquired optimal cutting point was applied on the test set to validate the training data. Using the threshold determined, vector maps with two habitat classes were created. Lastly, the acquired map was subset with management category map layers and proportions of *suitable* and *unsuitable* habitat were calculated for each category.

In study **II** the comparison approach with random points was used to investigate wader habitat selection. Distances between bird territories and random points to habitat features were measured. Next, a univariate logistic regression analysis was applied to measure and test the effect of study variables on nesting probability. After that, the ROC curve analysis was performed to estimate the prediction accuracy, sensitivity, and specificity distinguishing breeding sites and random points. As a more complex analysis, the multivariate logistic regression models followed by ROC curve analysis were fitted considering all study variables simultaneously. Also, random forest analysis was applied to find out the overall prediction accuracy of all variables together and to estimate the relative importance of variables.

In studies **III**, **IV,** and **V** nest survival analysis was carried out with MARK programme's RMark interface (Dinsmore & Dinsmore, 2007; Laake, 2011) and nest survival was expressed as daily survival rates (DSR). In study **III** distance to forest, distance to trees, forest cover (proportion of forest within a 1 km buffer around each nest), date of clutch initiation (calculated back based on the incubation stage; given in days from the 1st of March each year), year, species, and the interaction between distance to forest and species, were included in the model as covariates. Models that tested the effect of the year on distance to forest, distance to trees and forest cover were also included. To test the variation of forest cover and distance to trees and forest between the years, as well as to study nest predation events in relation to forest edge and forest cover by different species in study **III**, the Kruskal-Wallis test was used. Wilcoxon matched pairs test was used to compare red fox observations in active and inactive nests. To measure the effect of forest cover, distance to forest, and year on predator detection rates, these variables were used in a generalized linear mixed model using template model builder (glmmTMB) in the lme4 package. The same function and package were used in study **IV** to test the effect of the golden jackal on the red fox detection rate. In the nest survival analysis for study **IV**, the region was used as a variable to test if the presence or absence of the golden jackal could explain the variation in DSRs. As it was shown in study **III** that forest cover within a 1 km buffer around each nest has a negative effect on nest survival, and forest cover percentage around each nest is higher on island (mean=0.15, range=0.03–0.45) compared to mainland (mean=0.10, range=0.00–0.50; t-test results $t=5.21$, $df=655.23$, $p<0.001$), forest cover was included as a covariate in the survival model to account for this difference between the regions. Another covariate of interest was the year, as it is widely acknowledged that ecological processes vary over the years. In study **V** the survival and predators of artificial and natural nests were compared. Covariates included in the RMark model were camera presence, nest type, interaction of camera presence and nest type. In studies **III**, **IV**, and **V** the model with the lowest AIC_c value was considered to be the best model. All statistical analyses were performed using R (R Core Team 2018, 2020). A more thorough description of the statistical analyses used can be found in the Analysis or Data analysis subsections of the respective studies (**I–V**).

3. RESULTS

3.1. The agri-environment scheme supports breeding waders

Radar satellite data can be used to assess the quality of coastal grasslands by classifying vegetation height into two broad classes, *suitable* and *unsuitable*. The radar backscatter values of the two habitat categories formed two clear classes in the training set ($t = 9.22$, $df = 75.23$, $p\text{-value} < 0.001$). The habitat classification based on the decision boundary classified 88% of the *suitable* and 74% of the *unsuitable* values correctly on the test set. In addition, locations of 268 nests of eight wader species were used to verify habitat classification with nest locations resulting in 98% of the nests located in areas classified as *suitable* (Fig. 9)

In terms of vegetation height, 43% (9417 ha) of all Boreal Baltic coastal grasslands are in a favourable state for breeding waders in Estonia. Importantly, areas under an AES management achieve higher proportion of *suitable* habitat (60%) (Fig. 9, Fig. 10) and the largest proportion of *suitable* habitat (76%) is attained in areas receiving the higher tier conservation subsidy. Long-term grazing ensures better habitat quality, as in areas where the SNC subsidy has been paid for at least the last five years, 65% of the habitat has been classified as *suitable* (Fig. 9). In those areas where no management subsidies are paid only 24% of habitat was *suitable*. A more detailed presentation of the results can be found in the Results section of study **I**.

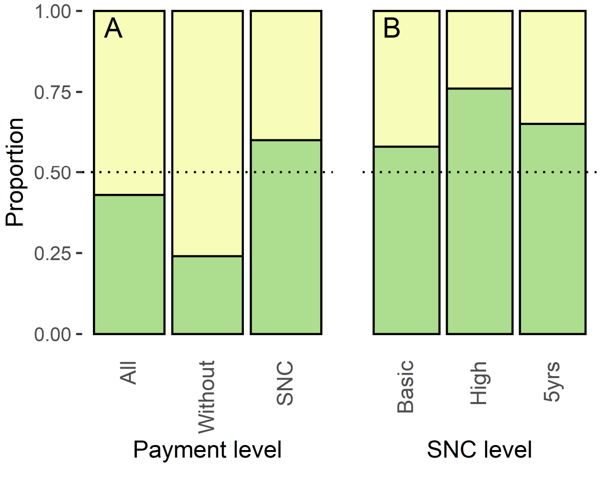

Figure 9. Proportion of the suitable habitat class (green bars) by (A) payment level, where *All* includes the entire coastal grassland habitat type, *Without* covers areas without AES subsidies and *SNC* are areas under the AES subsidies; (B) different semi-natural community subsidy levels, where *Basic* is the general SNC subsidy, *High* stands for the higher tier scheme, and *5yrs* includes areas that in 2019 had been under management for five or more years.

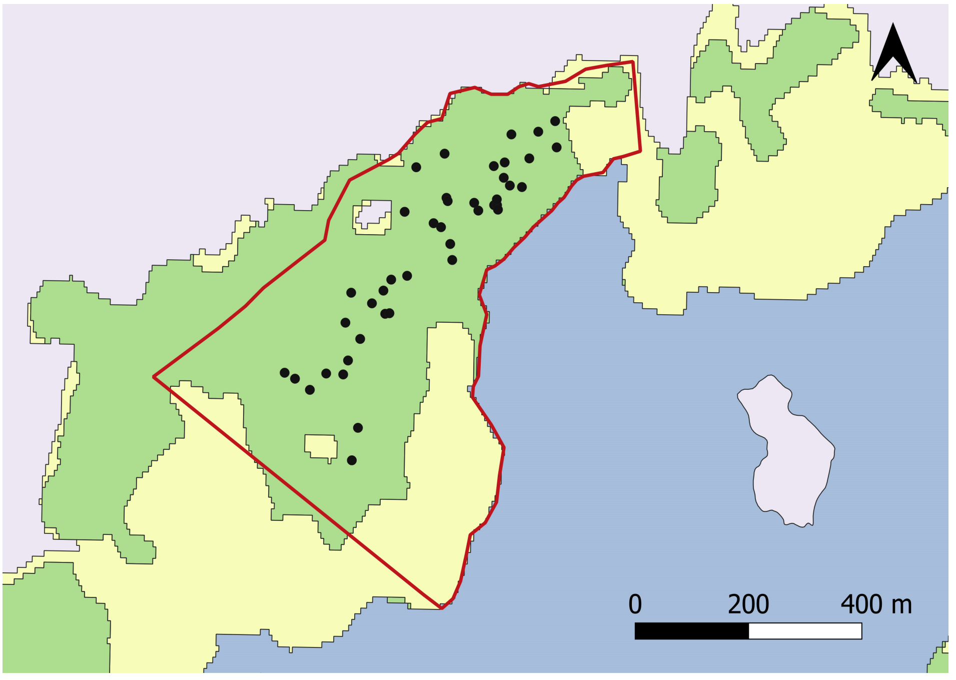

Figure 10. An example of coastal grassland habitat classification into *suitable* (green) and *unsuitable* (yellow) classes. Blue marks sea, grey land outside of coastal grassland habitat, the red polygon depicts the nest search area and black dots represent locations of wader nests found within this polygon.

3.2. Waders breed closer to wet areas and further from the forest

In two years, 864 wader breeding territories were mapped on 23 coastal grasslands. Results of logistic regression analyses showed that breeding sites of all studied species were closer to wet areas and coastline and further away from woodland than random points. Additionally, waders nested in lower areas of these grasslands than compared to random points.

The optimal thresholds distinguishing breeding sites from random points were similar for all studied species, indicating that waders chose the territory with higher probability if: (i) the distance from woodland was at least 100–200 m; (ii) the distance to wet areas was up to 20–50 m; (iii) the distance to coastline was up to 130–190 m; and (iv) the height of breeding ground was around 0.8 m (Fig. 11). However, the prediction accuracy of single variables remained weak for all species and variables (all AUC $\leq$ 0.6).

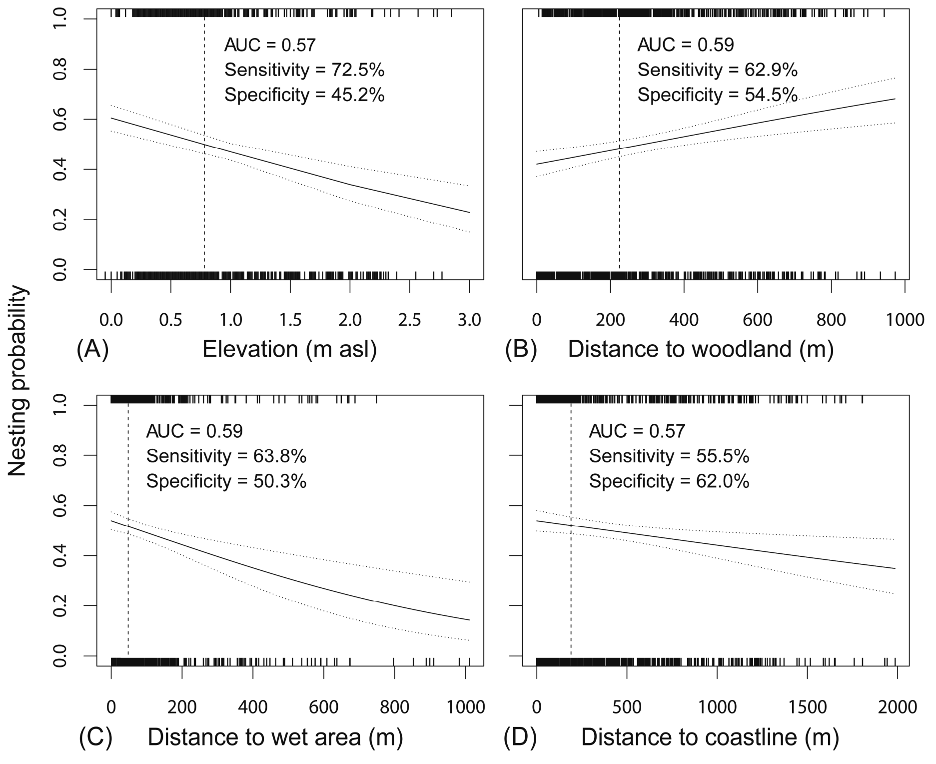

Figure 11. The southern dunlin's breeding probability depending on height of ground level (A) and distances from woodland (B), wet area (C) and coastline (D). Short vertical lines at each figure edges denote the empirical observations (breeding sites in upper edge and random points in lower edge); black solid line denotes the expected breeding probability (with 95% confidence interval) according to logistic regression; vertical dotted line denotes the optimal threshold corresponding to the maximum sum of sensitivity and specificity; sensitivity, specificity, and area under the ROC-curve (AUC) are presented numerically.

As the southern dunlin is a threatened species in the region its requirements for territory selection were examined in more detail. Southern dunlin nests were significantly further from the woodland compared to random points at the same sites and years ($p<0.001$) (Fig. 12). Furthermore, random points in areas with the breeding southern dunlin were further away from the woodland compared to random points at sites where the species did not breed (Fig. 12). This result indicates that one reason why the southern dunlin did not nest on some studied sites was the absence of suitable breeding areas far enough from the woodland edge. More detailed results are presented in the Results section of study **II**.

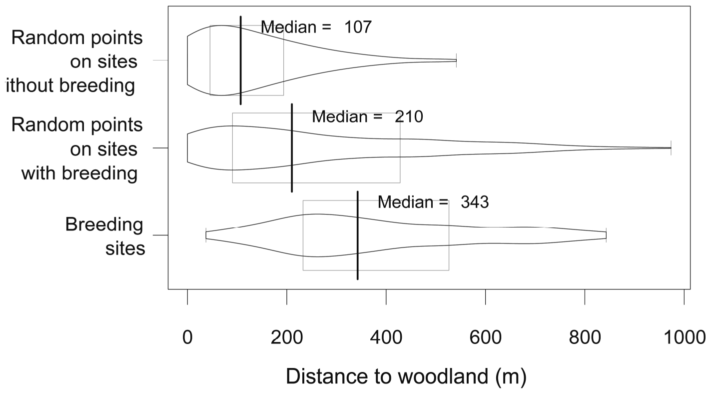

Figure 12. Distribution of distances to woodland at random points on sites where the southern dunlin does not breed, at random points on sites where the species breeds and at southern dunlin's territories presented as violin plots. Grey squares denote the inter-quartile range, strong vertical lines denote medians.

3.3. Nest predation by the red fox decreases wader nest survival close to the forest edge

During three breeding seasons 753 nests of 10 wader species were monitored: northern lapwing (n=444), common ringed plover (n=139), common redshank (n=91), southern dunlin (n=54), black-tailed godwit (n=8), ruff (n=5), common snipe (n=5), Eurasian oystercatcher (n=5), Eurasian curlew (n=1), and little ringed plover (n=1). Of these, 213 nests were successful, 526 failed, and in 14 cases the outcome was unknown. The majority of the nests that did not hatch were depredated (89%), while other causes of hatching failures included abandonment (6.7%), flooding (2.3%), trampling (0.2%), and an unknown reason (2.3%). The final analysis was based on 679 nests that were either successful or lost to predation. This dataset was used in both study **III** and **IV**.

For study **III**, the overall DSR calculated based on the null model was 0.929 (SE=0.0032, confidence interval (CI)=0.922–0.935). Top two MARK models explaining variance in nest survival differed <2 AIC_c units and included covariates: (i) forest cover and year, and (ii) distance to forest and year (Table 1A). Based on the 95% confidence intervals of beta coefficients that did not overlap zero, increasing distance to forest and decreasing forest cover had a positive effect on nest survival (Table 1B, Fig. 13).

Table 1. (A) Top two MARK models explaining the variation in daily survival rates, evaluating the effect of ForestCover, Forest, and Year; the full list of model results can be found in study **III**. (B) Beta estimates, standard error (SE), lower 95% confidence limit (LCL), and upper 95% confidence limit (UCL) for the best daily survival model. Year estimates are in relation to 2018. Predictors marked in bold have confidence intervals that do not overlap zero.

(A)

Model	Parameters	AICc	ΔAIC_c	Weight	Deviance
ForestCover + Year	**4**	**1457.89**	**0**	**0.63**	**1449.88**
Forest + Year	**4**	**1458.93**	**1.05**	**0.37**	**1450.93**

(B) Beta parameters

Model	Parameter	Estimate	SE	LCL	UCL
ForestCover + Year	Intercept	2.9263	0.1203	2.6906	3.1621
	ForestCover	**–1.6587**	**0.4069**	**–2.4563**	**–0.8612**
	Year2019	0.0156	0.1331	–0.2453	0.2765
	Year2020	**–0.4081**	**0.1282**	**–0.6595**	**–0.1568**
Forest + Year	Intercept	2.3942	0.1274	2.1445	2.6439
	Forest	**0.0007**	**0.0002**	**0.0003**	**0.0011**
	Year2019	0.0434	0.1328	–0.2168	0.3037
	Year2020	**–0.3710**	**0.1271**	**–0.6201**	**–0.1218**

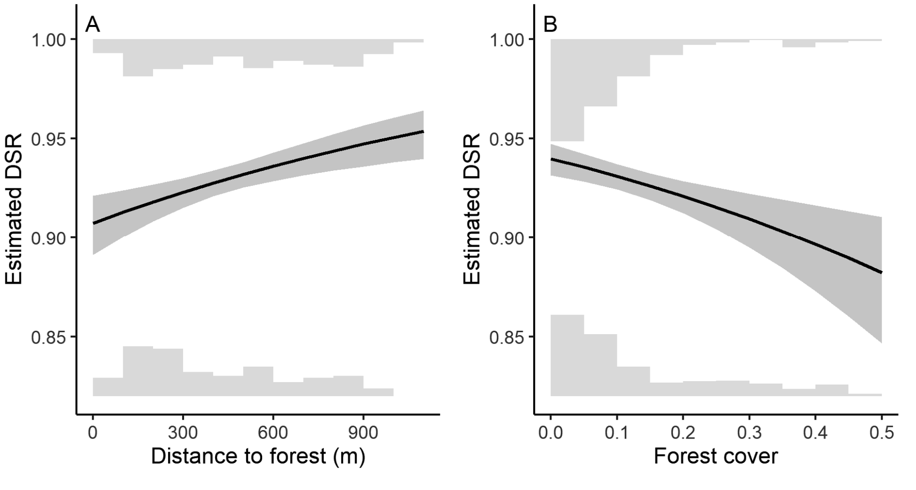

Figure 13. Estimated DSR in relation to distance to forest (A) and to proportion of forest cover within a 1 km buffer around each nest (B), the black line shows the predicted DSR and the grey area around it represents 95% confidence intervals (DSR predictions are based on models including only forest or forest cover). Histogram shows the distribution of the data, with hatched nests depicted at the top and depredated nests at the bottom of the plot.

Camera traps were used on a subset of 85 nests. Sixty-four of the nests with cameras were depredated. The red fox depredated 48%, golden jackal 8%, raven (*Corvus corax*) 5%, and badger (*Meles meles*) 3% of the nests, while in 36% of the predation events the camera malfunctioned and failed to record the nest predator. Additional potential predator species recorded by the cameras but not depredating nests were the raccoon dog (four observations) and the hooded crow (*Corvus cornix*; 77 observations). Predation by the red fox occurred on average at 217 m (95% CI=161–273 m) from the forest edge (Fig. 14). Based on very limited sample sizes, these results indicate that the northern raven also depredated nests close to the forest (mean distance=181 m (range=119–241 m)) while the European badger and golden jackal took nests further from the forest edge, at 606 (416–795 m) and 448 m (218–678 m), respectively. A comprehensive description of the results is given in the Results section of study **III.**

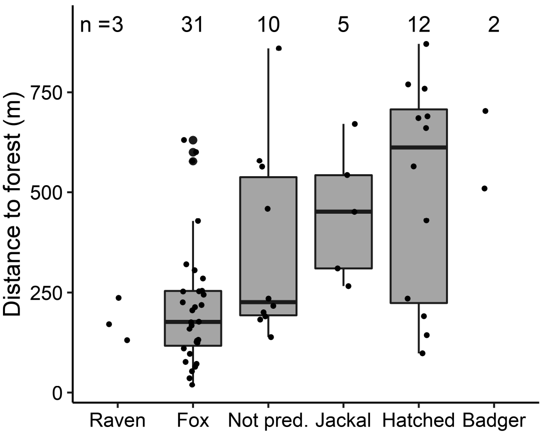

Figure 14. Nests depredated by different species, hatched nests, and nests that were not depredated while the camera was following the nest (cameras were removed from sites when cattle was released) in relation to their distance to forest. Grey boxes indicate the interquartile range, horizontal line within the box denotes the mean value, whiskers show the minimum or maximum value of the data that is 1.5 times the interquartile range over the 75th percentile while large circles indicate outliers. Small black dots present data points. For species with less than 5 observations, only data points are presented.

3.4. Nest predation by the golden jackal is compensatory

On the island study sites, the red fox was the only nest predator, while on the mainland sites the red fox and the golden jackal depredated the same proportion of nests. Additionally, on mainland the European badger depredated nests on one site and the northern raven took nests on two sites. The MARK model best explaining the variability in DSR included the region, forest cover, and year and accounted for 77% of model weight. Based on the 95% confidence intervals of beta coefficients that did not overlap zero, the region (island), forest cover and year (2020) have a negative effect on nest survival rates. This means that nest survival is higher in the area with the golden jackal. Nevertheless, the difference is marginal, with 95% confidence intervals for the real values overlapping, furthermore, nest survival rates are extremely low in both study areas (Fig. 15). Daily survival rates for both regions over all study years, considering the forest cover differences between the two regions, give the DSR value of 0.921 (95% CI 0.910–0.931) for the island and 0.934 (95% CI 0.926–0.942) for the mainland (calculated based on the Region+ForestCover model). If these values are presented as the proportion of nests that hatch after 27 days of laying and incubation, 11% (95% CI=8–15%) of the nests on the island and 16% (95% CI=13–20%) of the nests on the mainland reach hatching – both these values are most likely too low to sustain stable populations. Therefore, it can be concluded that the effect of the golden jackal is only compensatory.

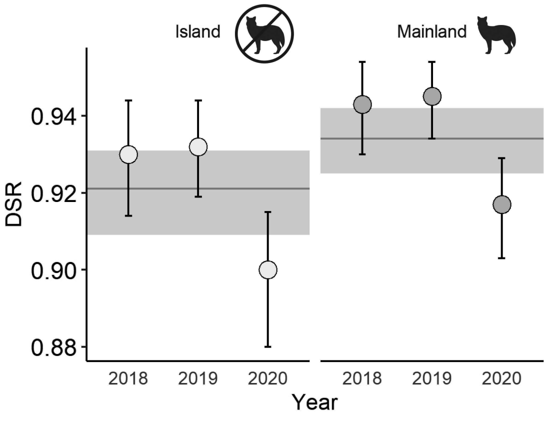

Figure 15. Daily survival rates (DSR) by region and year. Dots represent DSR values (based on the best model Region+ForestCover+Year) with 95% confidence intervals denoted as error bars. The grey area behind the dots represents the 95% confidence interval of the three-year overall DSR in the region, the grey horizontal line within the 95% confidence interval denotes the overall DSR (calculated based on the model Region+ ForestCover). Mainland represents the region with the golden jackal, island without the golden jackal.

Based on camera trap recordings, the detection rate of the red fox was 14 and 12 times higher on the island, compared to the mainland, in 2019 and 2020, respectively. Nevertheless, there was no evidence that the detection rate of the golden jackal would affect the detection rate of the red fox recorded by the same camera (GLMM results: Estimate=0.39, SE=0.39, p=0.33), however, the sample size for this test was small (red fox recorded in 41 cameras, golden jackal in 11). The full overview of the results of this study is presented in the Results section of study **IV**.

3.5. Artificial nests do not accurately reflect wader nest survival

A total of 60 artificial and 139 natural nests were monitored, of which 26 and 27 nests were equipped with a camera trap, respectively. Artificial nests without cameras (DSR=0.853, 95% CI=0.799–0.895) had a lower survival rate than natural nests without cameras (0.917, 95% CI=0.900–0.932, Fig. 16). The best model including covariates nest type, camera, and the interaction of these two, explained 77% of the model variation. Based on the model, the presence of the camera has an opposite effect on the two nest types, affecting artificial nest survival in a positive and natural nest survival in a negative direction.

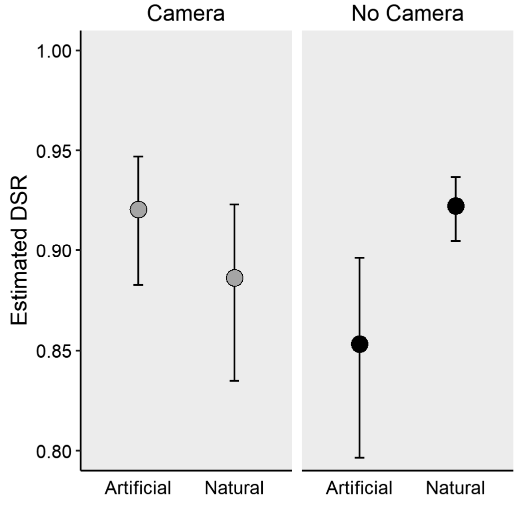

Figure 16. Estimated DSR values for artificial and natural nests with and without cameras; whiskers indicate 95% confidence intervals.

The red fox was the main predator of natural nests (40%) but shared the same proportion of nest predation events with the hooded crow at artificial nests (17%) (Fig. 17). The hooded crow was not identified depredating any natural nests. However, only weak evidence was found for different proportion of predator species between nest types ($X^2(5, n=12)=9.550$, $p=0.089$). In addition, three artificial nests with cameras were also trampled by cattle. A more detailed information on the results can be found in the Results of study **V**.

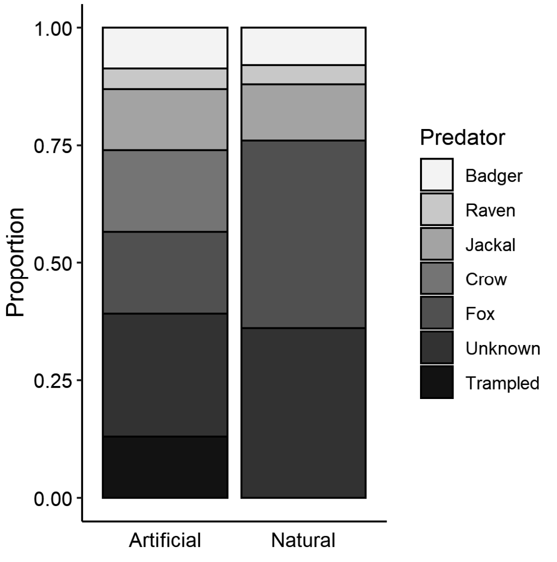

Figure 17. Proportions of predation events by different predators, and trampling by cattle, at artificial and natural nests. Artificial nests had 23 and natural nests 25 predation or trampling events. Three artificial and two natural nests reached the hatching date.

4. DISCUSSION

The thesis focuses on coastal grasslands, a type of protected semi-natural habitat endangered by overgrowth. The threatened breeding wader community is successfully used to explore habitat availability and selection, edge effect, as well as the impact of predators on ground-nesting birds (**I–V**). Many of the issues addressed in the thesis are driven by the need for effective evidence-based conservation measures. From a methodological perspective, the thesis advances the knowledge of the usefulness of radar satellite remote sensing in mapping the height of grassland vegetation (**I**). This methodology can be used to evaluate the condition of habitats from managed temperate grasslands to the Arctic tundra that are facing rapid agricultural or climate-induced shifts (Howison et al., 2018; Myers-Smith et al., 2011). Three of the studies in this thesis deal with the whole breeding wader community (**I**, **III**, **IV**). This rarely used community-wide approach helps to discern the needs of and threats to several species. These results help to lay the foundation for future research and conservation actions for semi-natural areas or wet grasslands inhabited by waders.

The dissertation revealed the extremely low survival rate of wader nests in all study sites (**III**), calling for urgent conservation action. These values are unprecedented, even among the worryingly low nest survival rates reported from other European countries (MacDonald & Bolton, 2008; Roodbergen et al., 2012), which have already been shown to contribute to species declines (Roos et al., 2018). Waders are long-lived species, with the average lifespan of the target species in this habitat, the dunlin (*Calidris alpina*), at 5.4 and maximum close to 29 years (Animal Diversity Web, 2022). For such species, high adult survival can mask poor breeding productivity for a long time before it can be detected in monitoring programmes (Roodbergen et al., 2012). Thus, we may soon be approaching the moment when these species pay their extinction debt (Kuussaari et al., 2009). Although this is still speculative and the idea needs to be proven using datasets that include other demographic parameters than hatching success over longer periods of time (Klok et al., 2009), it is a scenario that needs to be considered in order to take timely conservation actions. Otherwise, waders may follow the path of a once common, but now threatened amphibian that rapidly disappeared from these same coastal grasslands (Rannap et al., 2007). In the light of the probably unviable hatching success, it is important to understand the processes behind it, to propose mitigation measures, and to try to foresee what the future holds for these birds and habitat.

4.1. Key features of wader habitat

Birds have the ability to teach us about the state of the environment and their needs (Piersma & Lindström, 2004) and studying habitat selection is one way to “listen” to them. Waders select their territories close to wet features (pools or open shore) (Milsom et al., 2000; **II**) which are important foraging areas (Buchanan et al., 2006; Johansson & Blomqvist, 2010). They also breed in low areas (**II**) of these already low-lying coastal grasslands which may be driven by the preference for microhabitat in less windy locations or may be related to wetter soils that benefit their food objects. Currently, such habitat selection shows the relevance of the existence of wet features and heterogeneity in ground height, however, this knowledge may become particularly relevant under further climate change impacts. Overall drier climatic conditions during the wader breeding season are projected for the region (Kont et al., 2003). This may become a serious threat as dry conditions drastically reduce spatial and temporal availability of wet features and therefore impact feeding conditions of waders. It could be possible to mitigate these risks by restoring the wetness of the areas by closing ditches, dredging pools, and creating new wet features, but also keeping the coastline free of reed to favour foraging on the shoreline. Contrastingly, climate predictions also show more frequent storm surges (caused by strong westerlies) in combination with increased overall storminess and higher sea level (Kont et al., 2007). While this issue is likely to become more acute in the future, already during the 2020 breeding season several nests were washed away by two large-scale floods (**III**). This coincides with predictions from coastal wetlands in the United Kingdom (Ausden, 2014) and recent findings from the Bothnian Bay in Finland, where flooding of nests in coastal areas is already thought to be a major issue faced by the local southern dunlin breeding population (Pakanen et al., 2021). As the Boreal Baltic coastal grassland habitat is currently often a narrow strip between the sea and the woodland, it is important to understand the possibilities for restoring higher parts of coastal grasslands to extend their height range from the sea level. The two threats, low nest survival and flooding, also illustrate that a thorough understanding of the network of alternative wader breeding habitats, e.g., floodplain meadows, mires and arable land, and the knowledge on the connectivity between wader populations in these habitats, is important to obtain to buffer the population declines and possible loss of safe breeding habitat on coastal grasslands.

It is widely acknowledged that waders prefer open landscapes and disappear when habitats overgrow with higher plant species. The agri-environment scheme, the main tool for preserving biodiversity in the agricultural land in Europe (Kleijn & Sutherland, 2003), creates suitable habitat for waders in Estonian coastal grasslands (**I**). Although a specific agri-environment scheme is not directly comparable to schemes applied in different countries as the objectives and conditions are location-specific, it adds to the evidence that AESs can achieve their aims (Kleijn et al., 2006; Schekkerman et al., 2008; Smart et al., 2013), contrary to some other schemes with a similar focus (Mason et al., 2019). In the 2019 breeding season, ca 9400 ha of coastal grassland habitat was available for breeding waders in Estonia (**I**). This can be set as a baseline for future assessments, allowing to follow the trend of coastal grassland habitat availability for breeding waders.

In grazed areas, habitat management and nest survival is often seen as a trade-off between grassland preservation and nest trampling (Beintema & Muskens, 1987; Pakanen et al., 2011; Sharps et al., 2017). Fortunately, this is not the case in the studied areas where trampling rates are very low (**III**), promoting further grazing and increase in stocking rates to expand the area suitable for breeding waders (**I**). However, nest trampling is another topic that can become more pronounced in a changing climate. Currently, there is little for the livestock to eat in these grasslands before the end of May and low trampling rates are probably resulted from the relatively late release of cattle to the field (Pakanen et al., 2016; **III**). If the vegetation period moves forward, cattle will be released earlier. Unless the wader breeding phenology changes in parallel with the advance in vegetation growth, which is unlikely for all species (Meltofte et al., 2018), this will increase the temporal overlap between livestock grazing and wader breeding season, possibly leading to enhanced nest trampling rates (Pakanen et al., 2016). Constant monitoring of nest survival can help to register such change should it occur, and measures to limit cattle release time or stocking rates could then be considered.

4.2. Effects of habitat fragmentation

Contrary to habitat features that promote territory establishment, forest edge has often been identified as inhibiting waders (Hancock et al., 2009; Stroud et al., 1990). The edge effect question is a part of the larger topic of habitat fragmentation, which is thought to have a negative effect on biodiversity if little habitat is available (Rybicki et al., 2020, but see also Fahrig, 2003). As habitat fragmentation increases the edge length, it is one of the paths through which the effects of habitat fragmentation can operate. It has been suggested that habitat specialists suffer and generalist species benefit from the increase of edges (Henle et al., 2004). Waders as habitat specialists seem to obey this theory (Wilson et al., 2014; **II**; **III**). Albeit at different levels, the wader community breeding on coastal grasslands avoids breeding in the forest edge. Corroborating the results from the Scottish moorland (Hancock et al., 2009), the southern dunlin shows the strongest forest edge avoidance also on Estonian coastal grasslands, being even absent from sites narrower than a few hundred meters (**II**). Habitat loss and the resulting fragmented habitat is probably the leading reason behind its dramatic population decline (Elts et al., 2019; **II**). While the effect is weaker for other species that continue to breed in habitats fragmented with patches of woodland, they nevertheless avoid the forest edge, losing functional habitat at a much larger scale than just marked by the disappearance of suitable short vegetation. There might be several reasons for avoiding the forest edge, but in the context of our study system it is probably due to the elevated nest predation rates near the forest: only 7% of the nests made it to hatching in the proximity of the woodland (**III**). This finding gives rare quantitative support for the edge effect on nest survival in open habitats (Lahti, 2001).

When studying the edge effect on nest survival, it is essential to understand how predators use the habitat. Linking edge effects with identified predators has been a serious limitation in most studies, which largely build the conclusions upon a presumed list of suspects (Lahti, 2009). The reasons for this may lie in the difficulty of identifying nest predators, especially 30 or 40 years ago when most of the founding work on this topic was conducted (Paton, 1994). Even if predator identification was attempted, the methods used had limitations. For example, multiple studies investigating nest predation patterns in relation to habitat edge have been carried out using artificial nests (Major & Kendal, 1996), which can lead to distorted nest survival patterns and a misidentified suite of nest predators (MacIvor et al., 1990; Valkama et al., 1999; **V**). The advance of camera traps, which are generally regarded as an unbiased method (Mcguire et al., 2021; Salewski & Schmidt, 2020; **III**), has transformed the subject of predator identification. The use of this methodology allowed to identify the higher relative abundance and increased nest predation rates by the red fox closer to the woodland edge, suggesting this species as the cause behind the edge effect in this study system (**III**). As the red fox is a widespread predator, besides considered invasive in many regions (Global Invasive Species Database, 2022), this result may be directly applicable to other habitats.

The adverse forest edge effect on nest survival (**III**) is especially relevant in the current UN decade for ecosystem restoration (UNEP and FAO, 2020). In many parts of the world, waders nest in semi-natural habitats – areas where afforestation is often envisaged as the main restoration technique. Such processes have been undertaken most notably in the United Kingdom (Wilson et al., 2014), while also gaining popularity in Iceland (Gunnarsson, 2020). Our results imply that if the predator community includes species that benefit from the forest edge, effects of forest expansion will have negative consequences for breeding waders in the surrounding habitat. This is an important aspect to consider when designing forest restoration activities. As the Boreal Baltic coastal grassland is a priority habitat type mostly situated in protected areas in Estonia, it is currently largely safe from tree planting. Here the knowledge of an adverse edge effect supports the idea of reducing fragmentation of the coastal grassland habitat by removing recent patches of woodland as a possibly sustainable approach to tackling the issue of high nest predation rates.

4.3. Nest predators

Alterations in the habitat (Evans, 2004; Reino et al., 2010; **III**), eradication of rabies (Goszczyński et al., 2008), resource provisioning (Bino et al., 2010), and climate change (Spassov & Acosta-Pankov, 2019) have all possibly contributed to the increase of mammalian mesopredator populations (McMahon et al., 2020; Veeroja et al., 2021). In a situation where their prey species are already going through steep declines as a result of habitat loss, higher predation pressure exacerbates the complex of problems that can lead to local species extinctions (McMahon et al., 2020; Roos et al., 2018). Main wader nest predators have been suggested to be generalist mesopredators (McMahon et al., 2020), which is also the case in Estonian coastal grasslands, where most nests were taken by the red fox, followed by the golden jackal, the northern raven, and the European badger (**III**). Generalist species pose a further threat to their prey populations as they are independent of the abundance of a single prey species – after depleting one food resource, they can switch over to another one, without being affected by bottom-up control mechanisms. However, they can still be controlled by other processes, such as top-down regulation or competition with other predators (Ritchie & Johnson, 2009). The latter seems to be the case with the red fox whose detection rate is lower than expected in areas where it shares the landscape with the larger-bodied novel predator for the region, the golden jackal (**IV**). However, the presence of the golden jackal doesn't mean that nests are safe as this species also depredates nests (**IV**).

Habitat restoration has the potential to alleviate high nest predation rates, but it can be time-consuming (Kose et al., 2020; Sammul et al., 2012; **I**), thus additional methods to reduce high predator densities should be considered (Smith et al., 2010, 2011). In many areas where wader conservation is a priority, predator control is already implemented in management schemes as a normal practise (Franks et al., 2018), with varying but generally positive effect on nest survival (Bolton, et al., 2007; Neuman et al., 2004; Parr, 1993; Smith et al., 2010). For predator control to be effective and ethical, correct identification of nest predators prior to control actions is of high importance (**V**). For example, the use of artificial nests (e.g., Bravo et al., 2020; Olsen & Schmidt, 2004) can bias the predators towards corvids and should therefore be avoided (MacIvor et al., 1990; Valkama et al., 1999; **V**). A careful approach to predator control, with constant monitoring of the results to ensure that no predator species gains from the changing balance within the predator guild, is crucial.

Semi-natural grasslands, if managed suitably, provide a prime example of positive interaction between humans and nature, where food production helps to create and maintain a highly biodiverse habitat. Across Europe, farmland management is increasingly intensified and as a result, bird populations reliant on these open habitats are collapsing (Gregory et al., 2019). At the same time, semi-natural communities have been spared from intensification or land abandonment only on a fraction of their former extent. Nevertheless, these often small patches of habitat still hold high densities of open landscape species that are declining on intensive farmland, illustrating the stark contrast on wildlife between conventional intensive agriculture and the low-intensity livestock farming in semi-natural habitats (Tscharntke et al., 2012). Although the conservation of coastal grassland breeding waders is expected to become more complex, the significant value of semi-natural habitats is hopefully increasingly understood, resulting in more effective conservation actions to preserve the species dependent on these habitats.

5. CONCLUSIONS

In this thesis I investigated the wader perspective to Boreal Baltic coastal grasslands, focusing on topics related to habitat availability, breeding site selection, and nest predation. Based on the results of this work, the following can be concluded:

(I) The radar satellite remote sensing can be used to assess wet grassland habitat suitability for the breeding wader community in terms of vegetation height. Using this approach, I show that the agri-environment subsidy designed to benefit breeding waders on coastal grasslands has a positive effect on habitat suitability, although there is significant room for improvement. With the availability of the open-access remote sensing data, we can monitor habitat suitability of coastal grasslands, but possibly also other open habitats, with adequate temporal resolution and lay the basis for prompt conservation actions and improved management plans.

(II) Coastal grassland breeding waders avoid nesting close to forest edge, but prefer to breed closer to pools and coastline, while their habitat selection is not affected by the proximity to reedbed. These findings support restoration activities that aim to achieve large open areas at the expense of recent patches of woodland and emphasise the importance of keeping wet features accessible to waders.

(III) Habitat fragmentation impacts wader nest survival negatively through the adverse edge effect. The probable reason behind this pattern is the higher nest predation rate by the most common nest predator, the red fox, in the proximity of woodland. Such negative edge effect infers that the reason behind waders' forest avoidance may be the experienced higher predation risk. These results raise caution about the afforestation of open habitats in regions where nest predators benefit from forest cover. However, nest survival rates remain low even furthest from the forest, questioning the viability of the wader community breeding on coastal grasslands.

(IV) The predator community, the direct cause behind the alarmingly low nest survival rates, includes a novel predator for the region, the golden jackal. The effect of this species on the nests of ground-breeding birds is compensatory: while the golden jackal appears to reduce the presence of the red fox, it also depredates wader nests.

(V) The use of artificial clutches in nest survival studies of ground-breeding birds in an open habitat gives biased results. Thus, I reiterate the call for caution in studies using artificial nests to describe predation levels or explain predation patterns.

SUMMARY IN ESTONIAN

Pesitsevad kahlajad Läänemere rannaniitudel: elupaiga ulatus ja valik ning pesade ellujäämus

Elurikkuse praegune seisund maalib tumeda pildi inimkonna mõjust loodusele. Laialdase elurikkuse vaesumise taga on elupaikade kadu, mille peamisteks põhjusteks on ressursside ületarbimine ja üha intensiivistuv põllumajandus. Siiski võib inimese tegevus, sh põllumajandus, loodusele ka positiivselt mõjuda. Üheks selliseks näiteks on poollooduslikud ehk pärandkooslused, kus madala intensiivsusega põllumajandusvõtted, mis imiteerivad looduslikke häiringuid, on aastatuhandete jooksul ökosüsteeme kujundanud. Pärandkoosluste hulka kuuluvad mitmesugused elupaigad üle kogu maailma, karjatatavatest metsadest ja niidetavatest madalsoodest erinevat tüüpi püsirohumaadeni. Doktoritöö põhifookus on rannaniidul – elupaigal, mis on ühelt poolt pärandkooslus, teisalt aga märgala. Kuigi tegemist on ühe elupaigatüübiga, on töös käsitletavad teemad olulised kogu avatud märgade rohumaade kompleksile. Rannaniidud on vara-suktsessioonilised elupaigad, mis on tekkinud maakerke, mere mõju (üleujutused, jää) ning rändel olevate haneliste ja koduloomade karjatamise tulemusel. Rannaniidud pakuvad mitmesuguseid ökosüsteemi hüvesid, kindlustades muuhulgas paljudele inimestele sissetuleku ja heaolu. Rannaniitudel on ülioluline funktsioon ka elurikkuse hoidmisel, kuna seal kasvab umbes 30 kaitstavat taimeliiki, samuti on need elupaigaks kahepaiksetele ja mitmetele kaitsealustele linnuliikidele, seda nii pesitsus- kui rändepeatuspaigana.

Kuna rannaniitude taimestik on üldjuhul väheproduktiivne, on viimase 70 aasta jooksul põllumajanduses toimunud muutused soodustanud nende kasutusest kõrvalejätmist. Kui rannaniite ei karjatata, võtavad seal võimust suktsessiooniprotsessid, mille tagajärjel kaob elupaigale iseloomulik madal taimestik ja avatus. Lisaks vähendas rannaniitude pindala 1960. aastate puude istutamise kampaania „väheväärtuslike“ maade kasutusele võtmiseks. Kõige selle tagajärjel on rannaniitude pindala Eestis vähenenud hinnanguliselt 70%.

Koos elupaiga kaoga vaesuvad või hävivad ka rannaniitudel elavad linnupopulatsioonid. Linnurühm, kes sellest elupaigast enim sõltub, on kahlajad – peamiselt avatud märgaladega kohastunud liigid. Märgadel rohumaadel pesitsevad kahlajad on ohustatud üle kogu Euroopa ning kuigi asurkondade vähenemise põhjuseid on mitmeid, alates maakasutuse muutusest ja kuivendusest kuni väikekiskjate arvukuse suurenemiseni, on nende ühisnimetajaks inimmõju. Eesti rannaniitudel pesitsevad mitmed liigid, kes pole küll ülemaailmselt ohustatud, kuid kelle asurkonnad on langustrendis, nagu näiteks liivatüll (*Charadrius hiaticula*) või tikutaja (*Gallinago gallinago*). Lisaks on elupaik oluline ülemaailmselt ohulähedasteks hinnatud liikidele nagu kiivitaja (*Vanellus vanellus*), mustsaba-vigle (*Limosa limosa*), merisk (*Haematopus ostralegus*), suurkoovitaja (*Numenius arquata*) ja rohunepp (*Gallinago media*). Sihtliigid rannaniitude majandamisel on aga tutkas (*Calidris pugnax*) ja niidurüdi (*Calidris alpina schinzii*), kelle populatsioonid on peale drastilise languse Eestis vähenenud oluliselt ka Euroopa Liidus. Eestis kuuluvad need kaks liiki kõrgeimasse riiklikku kaitsekategooriasse. Kiivitaja ja tikutaja arvukus rannaniitudel on haudelinnustiku seire põhjal tõusutrendis, kuid kõikide teiste liikide arvukus on mõõdukas või tugevas languses. Kuna selline negatiivne trend kaasneb samaaegse elupaikade taastamise ja majandamisega, tekib küsimus, kas praeguseks killustunud rannaniidu maastikud on kahlajatele sobivaks elupaigaks. Seni on kahlajaid Eestis väga vähe uuritud, mis tähendab suuri teadmistelünki nende lindude langeva arvukuse põhjustes ja elupaiganõudluses. Doktoritöö eesmärk on neid teadmisi parandada. Alustasin laialdase, kogu elupaika hõlmava uuringuga ning seejärel keskendusin olulistele elupaiga tunnustele. Hindasin elupaiga killustatuse mõju pesade koorumisedukusele ning selgitasin välja pesarüüstajad. Viisin läbi ka väikesemahulise metoodilise uuringu tehispesade kasutamisest pesarüüste uurimisel.

Kuna suurt osa rannaniitudest majandatakse tänapäeval põllumajandus-keskkonnatoetuste abil, oli esimeseks ülesandeks hinnata selle looduskaitselise võtte mõju kahlajatele sobiliku elupaiga loomisel ja säilitamisel (**I**). Kogu elupaika hõlmava ülevaate saamiseks kasutasin kaugseire, täpsemalt Euroopa Kosmoseagentuuri radarsatelliidi Sentinel 1, andmeid. Rannaniitude majandamise tulemuslikkuse hindamiseks mõõtsin taimestiku kõrgust, kuna ala kõrge taimestikuga kinnikasvamine näitab rannaniidu kehva seisundit. Esmalt selgitasin välja, et radarsatelliidi andmete piiratuse tõttu saame taimestikku hinnata kahes kõrgusklassis, kahlajatele sobiv (<50 cm kõrgune) ja sobimatu (>50 cm kõrgune). Järgmisena vaatasin sobiva ja sobimatu elupaiga osakaalu hooldamata ja toetuste abil majandatud aladel. Kokkuvõttes selgus, et põllumajandus-keskkonnatoetusi saavate alade pindalast moodustas kahlajatele pesitsemiseks sobiv ala 60%, samas kui toetuseta piirkondades oli sobivat ala vaid 24%. Lisaks näitasin, et pikaajalisem karjatamine ja spetsiaalne kahlajatele suunatud toetussüsteem tagab parema tulemuse. Lisateadmisena leidsin, et hetkel on kahlajatele sobivat ala Eesti rannaniitudel kokku ca 9400 ha.

Kuigi majandatud rannaniitude pindala on viimase 20 aasta jooksul suurenenud, vajavad kahlajad lisaks madala rohustuga aladele ka teisi elupaiga omadusi, mis kindlustavad liikidele kvaliteetsed sigimis- ja toitumisvõimalused. Doktoritöö **II** osas uurisin, millised on kahlajatele olulised elupaiga tunnused taastatud, kuid siiski killustunud rohumaadel. Selgus, et kahlajate pesaterritooriumid asuvad lompidele ja avatud rannajoonele lähemal ning metsast kaugemal, kui eeldaks juhuslik jaotus. Lompide ja avatud ranniku olulisus pesitsusterritooriumite läheduses kinnitab varasemat teadmist, et kahlajatele on märjad kohad olulised, kuna need pakuvad rohkelt toitumisvõimalusi. Kuigi paljud rannaniidud on siiani osaliselt roostunud ja roostik kahlajatele elupaigaks ei sobi, kahlajad roostike lähedust iseenesest ei väldi. Rannaniitude majandamise seisukohast näitavad tulemused märgade kohtade avatuna hoidmise vajadust ning seda, et metsaga killustumine vähendab oluliselt kahlajate funktsionaalset elupaika. Ohustatud niidurüdi puhul näitasin, et metsa vältimine on liigil nii tugev, et kitsastel, alla paarisaja meetri laiustel, niitudel ta ei pesitsegi.

Elupaiga kaoga käib paratamatult kaasas selle killustumine. Üks killustumisega seotud mõjudest on servaefekt, mis linnupesade puhul võib tähendada kõrgemat pesarüüstet elupaiga servas. Kuigi selle protsessi toimimist on üsna hästi kirjeldatud metsaelupaigas, pole servaefekti negatiivne mõju avatud maastikes sageli kinnitust leidnud. Eesti rannaniidud, mis on praeguseks metsatukkadega killustunud avatud märgalad, pakuvad selle küsimuse uurimiseks hea mudelsüsteemi. Mida tähendab kahlajate jaoks elupaiga killustumine metsaga ja mis põhjustab metsaserva vältimist, selgitasin **III** uuringus. Töö raames viisin esimest korda Eestis läbi kahlajapesade koorumisedukuse uuringu, mille käigus jälgisin kolme aasta jooksul 753 pesa. Koorumisedukus on oluline demograafiline näitaja, mille põhjal saab hinnanguliselt kirjeldada populatsioonide elujõulisust. Uuring näitas ülimadalat kurnade koorumisedukust, mis viitab, et rannaniitudel pesitsevate kahlajate populatsioonid ei suuda ennast tõenäoliselt taastoota. Rüüstekoormus pesadele on kõrgeim metsa lähedal, kus koorub hinnanguliselt ainult 7% kurnadest; see on ka tõenäoline põhjus, miks kahlajad seda piirkonda pesitsemiseks väldivad (**II**). Kuigi koorumisedukus on kõrgem metsast kaugemal, jõuab ka seal koorumiseni vaid 26% kurnadest. Kui peamiselt piirdutakse servaefekti töödes oletustega pesarüüstaja liikide kohta, siis rajakaameraid kasutades selgus, et servaefekti põhjustab tõenäoliselt rebase kõrgem tihedus ja rüüstekoormus metsaserva lähedal. Need teadmised julgustavad rannaniitude killustatust vähendama ning sedakaudu kahlajate olukorda parandama. Samas näitavad tulemused, et väikekiskjate kõrge arvukus ohustab linnupesi ka metsast eemal, mistõttu on rannaniitudel pesitsevate kahlajapopulatsioonide säilimiseks oluline uurida ka teisi röövloomade arvukuse vähendamise võimalusi.

Röövloomade arvukuse reguleerimise mõjude uurimiseks on esmalt oluline teada, kes ja kui suure mõjuga pesi rüüstavad. Selgus, et enim ohustab kahlajapesi rebane (48%), kellel järgnesid šaakal (8%), ronk (5%) ja mäger (3%) (**III**). Šaakal on väikekiskja, kelle levila on viimase kümnendi jooksul oluliselt laienenud, kuid tema mõju saakloomadele hiljuti hõivatud aladel pole teada. Šaakali esinemine kõigil Eesti mandriosa uurimisaladel ning puudumine Hiiumaa aladel andis võimaluse seda teadmistelünka täita (**IV**). Võrdlevas uuringus näitasin, et kuigi šaakal rüüstab kahlajate pesi, tundub ta samal ajal vähendavat rebase esinemissagedust. Kuigi kurnade koorumisedukus oli šaakaliga aladel kõrgem, hindasin tema mõju siiski vaid kompenseerivaks, kuna ka šaakaliga aladel oli üldine koorumisedukus väga madal. Seetõttu ei paista šaakali esinemine koorumisedukusele ei negatiivset ega positiivset mõju avaldavat.

Koorumisedukuse mõõtmine on paljukasutatud meetod langeva arvukusega liikide halva käekäigu põhjuste diagnoosimisel ja looduskaitsemeetmete tõhususe hindamisel. Kuna sellised uuringud on töömahukad, on alternatiivina kasutatud linnupesi imiteerivaid tehispesi, mis uurija poolt elupaika paigutatakse ning mille saatuse põhjal pesarüüste kohta järeldusi tehakse. Kuna varasemad tehispesi kasutavad uuringud on näidanud, et see lähenemine võib viia eksitavatele tulemustele, on enne tehispesade kasutamist soovitatud nende vastavus pärispesadega kindlaks teha. Lootusega kasutada seda meetodit tulevikus Eesti rannaniitudel, hindasin tehispesade kasutamise võimalikkust (**V**). Näitasin, et tehispesade rüüste on kõrgem kui pärispesadel ning lisaks teistele liikidele rüüstavad neid ka varesed. Seega võib tehispesade kasutamine rüüstekoormust üle hinnata ning röövluskontrolli puhul viia valede otsusteni.

Doktoritöö tulemused on otseselt kasutatavad praktilises looduskaitses ning loovad pinnase tulevastele uuringutele ja tegevustele rannaniitudel. Tulemustest lähtuvalt on võimalik edaspidi uurida: a) kuidas peavad rannaniitudel maas pesitsevad linnud vastu kliimamuutusest põhjustatud sagenenud tormidele; b) kuidas mõjutavad kahlajaid, kui märgaladest sõltuvaid liike, tõenäoliselt sagenevad põuaperioodid pesitsusperioodil; c) milline on muude kahlajaelupaikade seisund ja kuivõrd on erinevate elupaikade populatsioonid omavahel seotud; d) kas suudame väikekiskjate arvukust vähendada? Kindlustamaks rannaniitudele iseloomulike kahlajapopulatsioonide säilimine, vajavad need ja paljud teised küsimused vastust juba lähitulevikus.

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Vansteelant, W. M. G., Wehrmann, J., Engelen, D., Jansen, J., Verhelst, B., Benjumea, R., Cavailles, S., **Kaasiku, T.**, Hoekstra, B., de Boer, F. (2020). Accounting for differential migration strategies between age groups to monitor raptor population dynamics in the eastern Black Sea flyway. *Ibis*, (162), 356–372. doi:10.1111/ibi.12773

Kaasiku, T., Rannap, R., & Kaart, T. (2019). Managing coastal grasslands for an endangered wader species can give positive results only when expanding the area of open landscape. *Journal for Nature Conservation*, 48, 12–19. doi:10.1016/j.jnc.2018.12.004

Conference presentations:

Kaasiku, T., Rannap, R. Millised on meie rannaniidud kahlajate elupaigana? (Coastal grasslands as wader breeding habitats) (oral presentation) Conference in celebration of the 100th anniversary of the Estonian Ornithological Society. 16.10.2021, Tartu, Estonia.

Kaasiku, T., Rannap, R., Männil, P. Elevated wader nest predation rates at a wet grassland-forest edge, (poster presentation) International Wader Study Group Annual Conference 8–10.10.2021, online.

Carneiro, C., Rocha, A.D., Nightingale, J., **Kaasiku, T.**, Rodrigues, M.S., Ferreira, H., Alves, J.A. Detailed habitat use by wintering waders at Portugal's largest and currently most threatened wetland, (poster presentation) International Wader Study Group Annual Conference, 8–10.10.2021, online.

Kaasiku, T., Rannap, R. Does a novel species in the predator guild affect wader nest survival? (Twitter presentation) International Shorebird Twitter Conference, 7–8.10.2020, online.

Kaasiku, T., Rannap, R. Does a novel species in the predator guild affect wader nest survival? (oral presentation) International Wader Study Group Annual Conference 10–12.10.2020, online.

Kaasiku, T., Praks J., Rannap, R. Assessing wader breeding habitat quality on a countrywide scale, using geotagged photos and satellite images, (oral presentation) International Wader Study Group Annual Conference 20–23.10.2019, Morecambe, UK.

Kaasiku, T., Vansteelant, W. Batumi Raptor Count: our strategy for monitoring more than one million raptors in the eastern Black Sea flyway, (oral presentation) International Bird Observatory Conference 28–31.03.2019, Eilat, Israel.

Kaasiku, T., Rannap, R. A conservation challenge: unsustainably high nest-predation rates of ground-breeding birds on coastal grasslands, (oral presentation) Botany and Zoology Conference 18.01.2019, Tartu, Estonia.

Pehlak, H., **Kaasiku, T.**, Kaldma, K., Luhamaa, H., Luigujõe, L., Sein, G., Tammekänd, I., Leibak, E., Leivits, A. First country-wide survey on breeding Jack Snipe in Estonia, (poster presentation) International Wader Study Group Annual Conference 28.09–01.10.2019, Workum, the Netherlands.

Kaasiku, T., Pehlak, H., Rannap, R. Limiting and supporting factors in breeding territory selection of meadow birds, (poster presentation) International Wader Study Group Annual Conference 15–18.09.2017, Prague, Czech Republic.

Dissertations supervised:

Helena Vanari, Bachelor's Degree, 2019. "Rannikumärgaladel pesitsevate kahlajate pesarüüste ohjamise võimaliikus ja viisid" (Nest predation management of waders nesting in the coastal wetlands). University of Tartu, Institute of Ecology and Earth Sciences, Department of Zoology.

Popular science publications:

Kaasiku, T., Rannap, R. (2020) Niidukahlajate arvukus kahaneb pesade suure rüüstekoormuse tõttu (High nest predation rates cause declines in wet grassland breeding waders). *Eesti Loodus*, 71(2), 54–61.

Hoesktra, B., Jansen, J., Engelen, D., de Boer, F., Benjumea, R., Wehrmann, J., Cavailles, S., **Kaasiku, T.**, Jansen, D., Fetting, P., Aintila, A., Vansteelant, W. (2020). Batumi Raptor Count: from migration counts to conservation in a raptor flyway under threat. *British Birds*, (113), 439–460.

Courses attended:

UAV Summer School, 3–5.08.2018, University of Tartu, Tartu, Estonia.

Adaptive Conservation Management Workshop, 15.–20.08.2018, CCNet Europe, Ikalto, Republic of Georgia.

Other activities and memberships:

Member of the Ornithological Society of the Middle East (since 2021)
Member of the International Wader Study Group (since 2013)
Member of the Estonian Ornithological Society (since 2012)

Voluntary work

Batumi Raptor Count volunteer raptor migration counter (since 2016), member of the monitoring team (since 2018)

Estonian Ornithological Society Rarities Committee additional member (since 2017)

International Wader Study Group conference coordinator (2015–2021)

Member of the International Wader Study Group executive committee (since 2015)

Volunteer at European Voluntary Service, lesser kestrel reintroduction programme, Almendralejo, Spain (August 2009 – June 2010)

Conference organisation

British Ornithologists' Union conference "Monitoring Science" 24–25.11.2021, online (technical assistance)

International Wader Study Group Annual Conference 8–10.10.2021, online (main organiser)

British Ornithologists' Union conference "Restoring Bird Populations" 30.03–01.04.2021, online (technical assistance)

International Wader Study Group Annual Conference 10–11.10.2020, online (main organiser)

British Ornithologists' Union and International Wader Study Group conference "International Shorebird Twitter Conference" 7–8.10.2020, online (co-organiser)

International Wader Study Group Annual Conference 26–29.09.2014, Haapsalu, Estonia (co-organiser)

Presentations at Environmental Board

Rannap, R., Kaasiku, T. Rannaniitude seisund (Status of coastal grasslands), (oral presentation) LIFE-IP infopäev, 09.06.2021, Kassari & 15.09.2021, Puise, Estonia.

Kaasiku, T., Rannap, R. Rannaniidud: seisund, lindude pesitsusedukus (Coastal grasslands: habitat status, nest survival), (oral presentation) Conservation Biology Day of Environmental Board, 14.11.2019, Tartu, Estonia.

ELULOOKIRJELDUS

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Haridustee:

2017– ...	Tartu Ülikool, zooloogia ja hüdrobioloogia, doktoriõpe
2015–2017	Tartu Ülikool, bioloogia, magistriõpe
2012–2015	Tartu Ülikool, bioloogia, bakalaureuseõpe
2007–2012	Kõrgem Kunstikool Pallas, mööbli restaureerimine, bakalaureuseõpe

Teenistuskäik:

2018– …	looduskaitsebioloogia spetsialist, Tartu Ülikool
2012–2017	projektijuht, OÜ Xenus

Peamised uurimisvaldkonnad: kahlajad, linnud, märgalad, pool-looduslikud kooslused, looduskaitsebioloogia

Teaduspublikatsioonid:

Kaasiku, T., Rannap, R., & Männil, P. (trükis). Predation-mediated edge effects reduce survival of wader nests at a wet grassland-forest edge. *Animal Conservation*. http://doi.org/10.1111/acv.12774

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Kaasiku, T., Praks, J., Jakobson, K., & Rannap, R. (2021). Radar remote sensing as a novel tool to assess the performance of an agri-environment scheme in coastal grasslands. *Basic and Applied Ecology*, 56, 464–475. doi:10.1016/j.baae.2021.07.002

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Konverentsiettekanded:

Kaasiku, T., Rannap, R. Millised on meie rannaniidud kahlajate elupaigana? (suuline ettekanne) Eesti ornitoloogiaühingu 100. aastapäeva konverents. 16.10.2021, Tartu, Eesti.

Kaasiku, T., Rannap, R., Männil, P. Elevated wader nest predation rates at a wet grassland-forest edge, (postriettekanne) International Wader Study Group Annual Conference 8–10.10.2021, veebis.

Carneiro, C., Rocha, A.D., Nightingale, J., **Kaasiku, T.**, Rodrigues, M.S., Ferreira, H., Alves, J.A. Detailed habitat use by wintering waders at Portugal's largest and currently most threatened wetland, (postriettekanne) International Wader Study Group Annual Conference 8–10.10.2021, veebis.

Kaasiku, T., Rannap, R. Does a novel species in the predator guild affect wader nest survival? (Twitteri ettekanne) International Shorebird Twitter Conference, 7.–8.10.2020, veebis.

Kaasiku, T., Rannap, R. Does a novel species in the predator guild affect wader nest survival? (suuline ettekanne) International Wader Study Group Annual Conference 10.–12.10.2020, veebis.

Kaasiku, T., Praks J., Rannap, R. Assessing wader breeding habitat quality on a countrywide scale, using geotagged photos and satellite images, (suuline ettekanne) International Wader Study Group Annual Conference 20.–23.10.2019, Morecambe, Suurbritannia.

Kaasiku, T., Vansteelant, W. Batumi Raptor Count: our strategy for monitoring more than one million raptors in the eastern Black Sea flyway, (suuline ettekanne) International Bird Observatory Conference 28–31.03.2019, Eilat, Israel.

Kaasiku, T., Rannap, R. A conservation challenge: unsustainably high nest-predation rates of ground-breeding birds on coastal grasslands, (suuline ettekanne) Botaanika ja Zooloogia konverents 18.01.2019, Tartu, Eesti.

Pehlak, H., **Kaasiku, T.**, Kaldma, K., Luhamaa, H., Luigujõe, L., Sein, G., Tammekänd, I., Leibak, E., Leivits, A. First country-wide survey on breeding Jack Snipe in Estonia, (postriettekanne) International Wader Study Group Annual Conference 28.09–01.10.2019, Workum, Holland.

Kaasiku, T., Pehlak, H., Rannap, R. Limiting and supporting factors in breeding territory selection of meadow birds, (postriettekanne) International Wader Study Group Annual Conference 15.–18.09.2017, Praha, Tšehhi.

Juhendatud väitekirjad:

Helena Vanari, bakalaureusekraad, 2019. "Rannikumärgaladel pesitsevate kahlajate pesarüüste ohjamise võimaliikus ja viisid". Tartu Ülikool, Ökoloogia ja Maateaduste Instituut, Zooloogia osakond.

Populaarteaduslikud publikatsioonid:

Kaasiku, T., Rannap, R. (2020) Niidukahlajate arvukus kahaneb pesade suure rüüstekoormuse tõttu. *Eesti Loodus*, 71(2), 54–61.

Hoesktra, B., Jansen, J., Engelen, D., de Boer, F., Benjumea, R., Wehrmann, J., Cavailles, S., **Kaasiku, T.**, Jansen, D., Fetting, P., Aintila, A., Vansteelant, W. (2020). Batumi Raptor Count: from migration counts to conservation in a raptor flyway under threat. *British Birds*, (113), 439–460.

Kursused:

UAV suvekool, 3–5.08.2018, Tartu Ülikool, Tartu, Eesti.

Adaptive Conservation Management Workshop, 15.–20.08.2018, CCNet Europe, Ikalto, Gruusia.

Muud tegevused ja liikmelisus:

Ornithological Society of the Middle East liige (alates 2021)
International Wader Study Group liige (alates 2013)
Eesti ornitoloogiaühingu liige (alates 2012)

Vabatahtlik töö

Batumi Raptor Count röövlindude rände loendaja (alates 2016), seiretiimi liige (alates 2018)

Eesti ornitoloogiaühingu harulduste komisjoni varuliige (alates 2017)

International Wader Study Group konverentside koordinaator (2015–2021)

International Wader Study Group juhtkogu liige (alates 2015)

Euroopa Vabatahtlik Teenistus, stepi-tuuletallaja taasasustamise projekt, Almendralejo, Hispaania (august 2009–juuni 2010)

Konverentside korraldamine

British Ornithologists' Union konverents "Monitoring Science" 24–25.11.2021, veebis (tehniline tugi)

International Wader Study Group Annual Conference 8–10.10.2021, veebis (korraldusmeeskonna juht)

British Ornithologists' Union konverents "Restoring Bird Populations" 30.03–01.04.2021, veebis (tehniline tugi)

International Wader Study Group Annual Conference 10–11.10.2020, veebis (korraldusmeeskonna juht)

British Ornithologists' Union and International Wader Study Group konverents "International Shorebird Twitter Conference" 7–8.10.2020, veebis (korraldusmeeskonna liige)

International Wader Study Group Annual Conference 26–29.09.2014, Haapsalu, Eesti (korraldusmeeskonna liige)

Esitlused Keskkonnaametis

Rannap, R., Kaasiku, T. Rannaniitude seisund, LIFE-IP infopäev, 09.06.2021, Kassari ja 15.09.2021, Puise, Eesti.

Kaasiku, T., Rannap, R. Rannaniidud: seisund, lindude pesitsusedukus, Keskkonnaameti looduskaitsebioloogia koolitus, 14.11.2019, Tartu, Eesti.

DISSERTATIONES BIOLOGICAE UNIVERSITATIS TARTUENSIS

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