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# Monitoring Nest Predation in Ground-Nesting Birds in the Low-Arctic Tundra - a Test of Methods

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Masters in Ecology

# Acknowledgments

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# Abstract

Ground-nesting birds in the Low Arctic tundra are experiencing population declines, with increasing nest predation playing a critical role. Due to climate change, meso-predators like the red fox (*Vulpes vulpes*) and corvids (*Corvidae* spp.), such as the hooded crow (*Corvus cornix*) and common raven (*Corvus corax*), have become more abundant in the changing tundra biome. These generalist predators respond numerically to the influx of biomass in the years with high rodent abundances, hence their populations rise during rodent peak years. The year following a rodent population peak is often referred to as the crash year. During this period, predator numbers remain high and may result in increased predation on ground-nesting birds, which generally lack effective natural defenses.

Studying nest predation in the tundra presents challenges, as conventional camera monitoring methods with visible camera objects are known to impact predation rates. Additionally, former studies have highlighted the differences between artificial nest setups and real nests. Thus, a more discrete monitoring method that can be used on real nests is needed to obtain a more representative predation rate. This study investigates whether smaller and well-hidden cameras influence predation rates on artificial ground nests in the tundra. The more compact and less visible cameras used in this study were hypothesized to not affect predation rates. Furthermore, the predation rates of artificial nests were compared with real nests to assess the validity of artificial nest experiments. Lastly, yearly data on rodent abundance and predation in artificial nests across three locations were analyzed to identify the drivers of interannual variation in predation pressure.

Results indicate that smaller and concealed cameras reduced bias in measuring predation pressure on artificial nests, compared to traditional methods. Additionally, predation probability did not differ significantly between artificial nests monitored by a concealed camera and the control nests without a camera. Furthermore, the comparison of predation probability between artificial nests and two different categories of real nests indicated an overall difference in predation probability among the three groups. However, the comparison showed that artificial nests more closely resembled those of shorebirds (order *Charadriiformes*) and willow ptarmigan (*Lagopus lagopus*) than to those of arctic-alpine passerines (order *Passeriformes)*, in terms of both predation pressure and appearance. Lastly, the analysis of rodent abundance and artificial nest predation relationship showed that nest predation increased the year after a peak rodent year.

These findings support the development of an unbiased camera monitoring method for real nests, enabling accurate identification of nest predators and true predation rates. Long-term monitoring across all stages of the rodent cycle is necessary, as fluctuations in rodent populations directly impact predation pressure on bird nests. Identifying key predators and quantifying predation rates will allow for targeted conservation measures to prevent further declines in ground-nesting birds in the Low Arctic.

# Sammendrag

Bakkehekkende fugler på den lavarktiske tundraen opplever en bestandsnedgang, hvor økt reirpredasjon spiller en kritisk rolle. På grunn av klimaendringer har mellomstore predatorer som rødrev (*Vulpes vulpes*) og kråkefugler (*Corvidae* spp.), som kråke (*Corvus cornix*) og ravn (*Corvus corax*), blitt mer tallrike i et tundrabiom i rask endring. Disse generalistene responderer numerisk på gode smågnagerebestander, noe som fører til en større populasjonsstørrelse etter smågnager-toppår. I det påfølgende smågnager-krasjåret kan den økte rovdyrbestanden skifte fokuset sitt til sårbare bakkehekkende fuglereir. Disse fuglene har utviklet få forsvar mot disse nye predatorene.

Reirpredasjon på tundraen er et utfordrende studium ettersom de nåværende overvåkingsmetodene, som innebærer synlige kamerastrukturer, er kjent for å påvirke predasjonsraten. Videre har tidligere studier belyst flere viktige forskjeller mellom kunstige- og virkelige reir. Derfor er det behov for en mer diskret overvåkingsmetode som kan brukes på ekte reir, for å få en mer representativ predasjonsrate. Denne studien undersøker hvorvidt en godt skjult kameraovervåkingsmetode påvirker predasjonsraten på kunstige bakkereir på tundraen. De mer kompakte og lite synlige kameraene som ble tatt i bruk i denne studien ble antatt å ikke påvirke predasjonsraten. Videre ble predasjonsraten for de kunstige reirene sammenlignet med de virkelige reirene for å vurdere relevansen til kunstige reireksperimenter. Til slutt ble årlig data på smågnagerbestanden og reirpredasjon på kunstige reir i tre forskjellige områder analysert for å undersøke hva som driver mellomårsvariasjonen i predasjonsraten på kunstige reir.

Resultatene indikerte at mindre og skjulte kameraer gav et mer representativt bilde av predasjonspress på kunstige reir, sammenlignet med tidligere metoder. I tillegg var ikke predasjonssannsynligheten på de kunstige reirene med skjult kameraovervåking signifikant forskjellig fra kontrollreirene uten kameraovervåking. Videre viste sammenligningen av sannsynligheten for predasjon mellom kunstige reir og to ulike kategorier av virkelige reir at det var en overordnet forskjell i predasjonssannsynligheten mellom de tre gruppene. Derimot viste sammenligningen at kunstige reir minner mer om reirene til vadefugler (order *Charadriiformes*) og lirype (*Lagopus lagopus*) enn reirene til de arktisk-alpine spurvefuglene (order

*Passeriformes*), både når det gjelder predasjonspress og utseende. Til slutt viste analysen av forholdet mellom smågnagerpopulasjonsstørrelse og predasjon på kunstige reir at predasjonen på reir økte året etter en topp i gnagerbestanden, selv om det her var betydelig romlig variasjon.

Disse funnene er et viktig steg på veien i utviklingen av en kameraovervåkingsmetode som ikke påvirker predasjonsraten på virkelige reir, hvor man kan identifisere reirpredatorene og estimere den reelle reirpredasjonsraten. Langsiktig overvåking av reirpredasjon gjennom alle fasene av smågnagersyklusen er viktig, da fluktuasjonene i bestandsstørrelsen hos smågnagere kan gi ulikt predasjonspress på fuglereirene. Ved å identifisere predatorene, i tillegg til å fastslå predasjonsraten, får man et godt kunnskapsgrunnlag til å kunne iverksette bevaringstiltak for å forhindre ytterligere nedgang i bestanden av bakkehekkende fugl i lavarktis.

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# 1. Introduction

Climatic changes are among the primary drivers of biodiversity loss worldwide (Field & Barros, 2014; Stephens et al., 2016). Species are affected by altered abiotic factors induced by climate change, as well as the continuing reduction of habitat caused by increasing pressure from human land use (Fahrig, 2003). In both Arctic and alpine regions, climatic changes are amplified compared to those observed at lower latitudes and altitudes (Gonzalez et al., 2010; Rantanen et al., 2022). Such changes contribute to shorter snow cover seasons (Derksen & Mudryk, 2023). Rising temperatures and environmental changes are allowing generalist predators, not commonly found in the Arctic and alpine regions, to move beyond their historical latitudinal and altitudinal ranges (Elmhagen et al., 2015; Gallant et al., 2020; Sokolov et al., 2016). Additionally, extreme weather events, which are becoming more frequent due to climate change, contribute to reindeer mortality and, in turn, facilitate the movement of boreal predators into the Arctic (Sokolov et al., 2016). Arctic greening, a phenomenon driven by climate change, is increasing primary production in the Arctic, which has additionally been discovered to inflate the number of meso-predators and generalist consumers in the north (Ims et al., 2019; Phoenix & Treharne, 2022). Although this increased primary production can, in certain regions, be countered by insect outbreaks, which are also found to be more pronounced further north (Jepsen et al., 2008). Range shifts such as this shows that living organisms and ecosystems are responding to climate change (Thomas, 2010).

The meso-predator northwards expansion will likely increase the nest predation rate in the Arctic, as food web theory suggests (Hastings, 1988; Ims et al., 2019). This is bad news for the alpine and Arctic ground-nesting birds in Europe and North America, who already are in a decline (Johnston et al., 2025; Lehikoinen et al., 2019). These birds are an important and unique class of species, some of which are adapted to long migrations and extreme climatic conditions (Johnson & Herter, 1990). Since there are no trees, they are nesting on the ground – a vulnerable behavior (Hof et al., 2017; Reif et al., 2023; Roos et al., 2018) in the most fragile part of the bird's life cycle (Husby & Hoset, 2018). These birds are poorly adapted to the extent of meso-predators the future alpine and Arctic ecosystems are expected to hold (Hof et al., 2012; Ims et al., 2019). A harsh environment without too many predators is a decisive factor for birds deciding to nest in these low productivity areas in the first place (Callaghan et al., 2004; Kubelka et al., 2018; McKinnon et al., 2010). The combination of an increase in predators and the vulnerability in the birds is expected to have a cascading effect on the future abundance of the ground-nesting birds (Ims et al., 2019).

Nest predation rates are closely linked to rodent abundance (Bêty et al., 2001; Gauthier et al., 2004). Rodents such as the Norwegian lemming (*Lemmus lemmus*) and the gray-sided vole (*Myodes rufocanus*) are key stone species for the tundra ecosystem, as they are one of few primary consumers (Batzli et al., 1980; Pokrovsky et al., 2015). Adaptations to a nutrient poor environment have made them effective in turning energy from the primary producers to higher trophic levels (Batzli & Jung, 1980). The tundra has less biodiversity than other habitats, meaning fewer species occupy each trophic level (Ims & Fuglei, 2005). In other habitats, where there are more primary consumers, predators are not as dependent on one species as they are in the tundra (Legagneux et al., 2012). Rodent populations fluctuate in cycles where their numbers rise and fall over periods of three to five years (Andreassen et al., 2021; Myers & Krebs, 1974; Yoccoz & Ims, 2004). As one of the few prey species in the nutrient-poor tundra, cyclical outbreaks in small rodent populations drive a numerical response from predators (Angelstam et al., 1984; Batzli, 1983). During peak years, predators feed on the abundant rodent biomass, leading to increased reproduction (Angelstam et al., 1984). The year following the rodent population peak typically sees a sharp crash in rodent numbers, but predator populations remain high, sustained by the previous year's abundance of prey (Hansson & Henttonen, 1988). These years with few rodents and a high abundance of predators have a detrimental effect on alternative prey like ground-nesting birds (Bêty et al., 2001; Gauthier et al., 2004; Lehikoinen et al., 2016). This is referred to as the alternative prey hypothesis (Angelstam et al., 1984). For instance, a study by Marolla et al. (2019) on the Lesser white-fronted goose (*Anser erythropus*), in the western part of Finnmark, showed a 70% drop in probability of nesting success between years with rodent peaks and years with rodent crash. Former studies, like Munkebye et al. (2003) have looked at nest predation rates in the Norwegian alpine tundra without taking the state of the rodent population into consideration, providing an incomplete understanding of the different factors influencing nest predation.

Nest predation is having a huge impact on bird populations (Colwell, 2010). Predation on an entire clutch has a greater impact on the population than predation on a single individual (Kubelka et al., 2018). Offspring recruitment counters individual mortality and helps maintain sustainable bird populations, thus disruption of reproductive success can have damaging effects on populations (Kubelka et al., 2018). Nest predation is expected to increase in the Arctic due to both the expansion of nest predators into the region (Ims et al., 2019; Kubelka et al., 2018) and cascading ecological effects, such as the collapse of small rodent population cycles (Ims et al., 2008; Kausrud et al., 2008). Increased snow hardness linked to rain on snow and temperature fluctuations above the freezing point in the winter destroys the insulating subnivean space, limiting both rodent movement and access to food (Ims et al., 2008; Kausrud et al., 2008). As a result, nest predators may increasingly rely on ground-nests as prey in the future (Lehikoinen et al., 2016). A huge portion of the species among the ground-nesting birds in the alpine and Arctic region are already red listed (Hof et al., 2017), yet there is limited information regarding which specific predator species contribute most to nest predation (Benson et al., 2010; Moore & Robinson, 2004; Thompson III & Burhans, 2004). Given these pressures, monitoring nest predation in the Arctic is more crucial than ever.

Since nest predation is a major factor limiting the abundance of ground-nesting birds and is expected to increase, it is crucial to both quantify predation rates (Helmstetter et al., 2024) and identify the specific predator species involved (Richardson et al., 2009; Thompson III, 2007). Artificial nests are widely used to estimate nest predation rates because they are non-intrusive and allow for experimental control (Bravo et al., 2020; Major & Kendal, 1996). Various methods, such as using real eggs like those from domestic hens or common quail (*Coturnix coturnix*), are employed to mimic the conditions of natural nests (Ims et al., 2019; Major & Kendal, 1996). However, while effective for assessing relative predation pressure, artificial nests do not provide information about predator identity on their own (McKinnon & Bêty, 2009; Thompson III & Burhans, 2004). Some studies have attempted to determine predator type using plasticine eggs, which can retain bite or beak marks, but this method generally only allows identification to broad groups, for instance distinguishing mammals from birds, rather than the specific species (Ims et al., 2019). Due to these limitations commercially available wildlife cameras are as of now the most used tool for monitoring of both nest predation rate and predator identification (Bravo et al., 2020;

Richardson et al., 2009). Studies with the use of artificial nests in combination with common wildlife cameras show however, that this method might lead to biased results (Bravo et al., 2020; Henden et al., 2024).

Predators commonly associated with artificial nest predation are corvids (Erikstad et al., 1982; Parker, 1984). Corvids have high cognitive abilities; they follow humans and spot novel objects like the cameras and therefore the nests (Sonerud & Fjeld, 1987). Even though former studies like Herranz et al. (2002), Richardson et al. (2009) and Bravo et al. (2020), have found a scaring effect from cameras in south, it seems like an opposite effect in the Low Arctic (Henden et al., 2024). An explanation for this regional difference might be that the corvids have been actively trapped in the southern and more human-dominated ecosystems found in USA and central-Europe, thus they may have learned that human-made objects pose as a threat (Henden et al., 2024). Although hunting also has occurred in the pristine north, the decline in natural resources may have led corvids to associate human objects with nutritional sources to a greater extent than in the south (Henden et al., 2024; Restani et al., 2001). It is also known that generalist predators like the red fox are attracted to human-objects such as garbage (Elmhagen et al., 2015; Gallant et al., 2020; Sokolov et al., 2016). In any case there seems to be a significant correlation, and bias between nest predation and the use of conspicuous wildlife cameras to monitor the nests in the tundra (Henden et al., 2024). The low vegetational height in the tundra makes concealment of camera equipment challenging (Henden et al., 2024). Thus, Henden et al. (2024) has called "for new technologies that allow for photographic monitoring of bird nests with minimal visual footprints". In addition, Herranz et al. (2002) recommended that future nest predation research use small, camouflaged cameras and include control nests without camera monitoring. This study will address this issue.

Lastly it is crucial to examine the relevance of results found by using artificial nests, as they may not apply directly to real nests (Pärt & Wretenberg, 2002; Willebrand & Marcström, 1988). To assume that predation on artificial nests automatically transfers to real nests would be naïve. There are several differences between artificial and real nests that needs to be acknowledged. The artificial nests are not defended by a parental bird. However, this bird does not reveal the nest's location, for instance, while foraging (Pärt & Wretenberg, 2002). In real nests, higher nest attendance during incubation is associated with reduced predation risk, potentially due to increased camouflage from the cryptic incubating bird (Meyer et al., 2020; Smith et al., 2012). Artificial nests, on the other hand, lie exposed and lack such concealment. Additionally, artificial nests do not take into consideration the birds' choice of nest habitat (Pärt & Wretenberg, 2002). Lastly, artificial nests set up by humans not only retain human scent but also have a significant visual footprint at the time of placement (Bravo et al., 2020). Former studies show that artificial nests have a higher predation rate than real nests (Bentzen et al., 2017; Burke et al., 2004; Major & Kendal, 1996). Furthermore, it seems that natural nests to a larger degree are depredated by mammalian predators, whereas artificial nests are more frequently targeted by birds (MaCivor et al., 1990; Willebrand & Marcström, 1988). Thus, studies using artificial nests might not provide an accurate identification of the actual predators of real nests nor a true estimate of the predation rate.

This thesis investigated whether a small and well camouflaged camera monitoring setup affected the predation rate of artificial nests less than the previously used, more conspicuous trail camera monitoring method, mounted on a pole (1). Additionally, it was tested whether the predation rate of artificial nests differed from the predation rate of real nests (2). Lastly, the correlation between predation rates on artificial nests and the number of rodents in the same area was studied (3).

It was hypothesized for (1) that the smaller, camouflaged camera method would result in a less biased estimate of nest predation rates than the previously used, more conspicuous camera method. Specifically, it was expected that nests monitored with the smaller camera would exhibit predation rates similar to those of control nests without camera monitoring. For (2) it was expected that the artificial nests would experience a higher predation rate than the real nests. Finally, for (3) it was predicted that the predation rate on artificial nests would increase in years following a high abundance rodent year. Some parts of this introduction are adapted from a term paper written for the preparatory master's course 'Methods in Natural Science' (Ekeli, 2024).

# 2. Materials and methods

## 2.1 Study area

This study was carried out on the Varanger Peninsula, in northeastern Norway (Figure 1). The study area had limited tree cover due to its location in the transition zone between the boreal forest and the Low Arctic tundra (Vindstad et al., 2017; Vindstad et al., 2019). Although situated in this ecotonal region, it was classified as a Low Arctic area at 70 degrees north (Siwertsson et al., 2023; Walker et al., 2005). The vegetation was low-growing and characterized by crowberry (*Empetrum hermaphroditum*), dwarf willow *(Salix herbacea*), bilberry (*Vaccinium myrtillus*), bog bilberry (*Vaccinium uliginosum*), and lingonberry (*Vaccinium vitis-idaea*), willow thickets, grasses, sedges, lichens, and mosses (Henden et al., 2011). Small mammal herbivores included the gray-sided vole, the tundra vole (*Microtus oeconomus*) and the Norwegian lemming (Ims et al., 2011). A significant portion of the diverse focal birdlife nested on the ground. The unique ground-nesting bird community consisted of Arctic-alpine passerines (e.g., Lapland longspur *Calcarius lapponicus* and meadow pipit *Anthus pratensis*), shorebirds (e.g., dunlin *Calidris alpina* and Eurasian Golden Plover *Pluvialis apricaria*), and galliform birds (order Galliformes, e.g., willow ptarmigan and rock ptarmigan *Lagopus muta*; Ims & Henden, 2012). Predators in the area included wolverines (*Gulo gulo*), red and Arctic foxes (*Vulpes vulpes* and *Vulpes lagopus*), stoats and weasels (*Mustela erminea* and *Mustela nivalis*), as well as White-tailed eagles (*Haliaeetus albicilla*), Golden eagles (*Aquila chrysaetos*), and gyrfalcons (*Falco rusticolus*). Of greater relevance to this thesis, the potential nest predators consisted of foxes (*Vulpes* spp.), corvids (*Corvidae* spp.), skuas (*Stercorariidae* spp.), and mustelids (*Mustelidae* spp.; Ims et al., 2017).

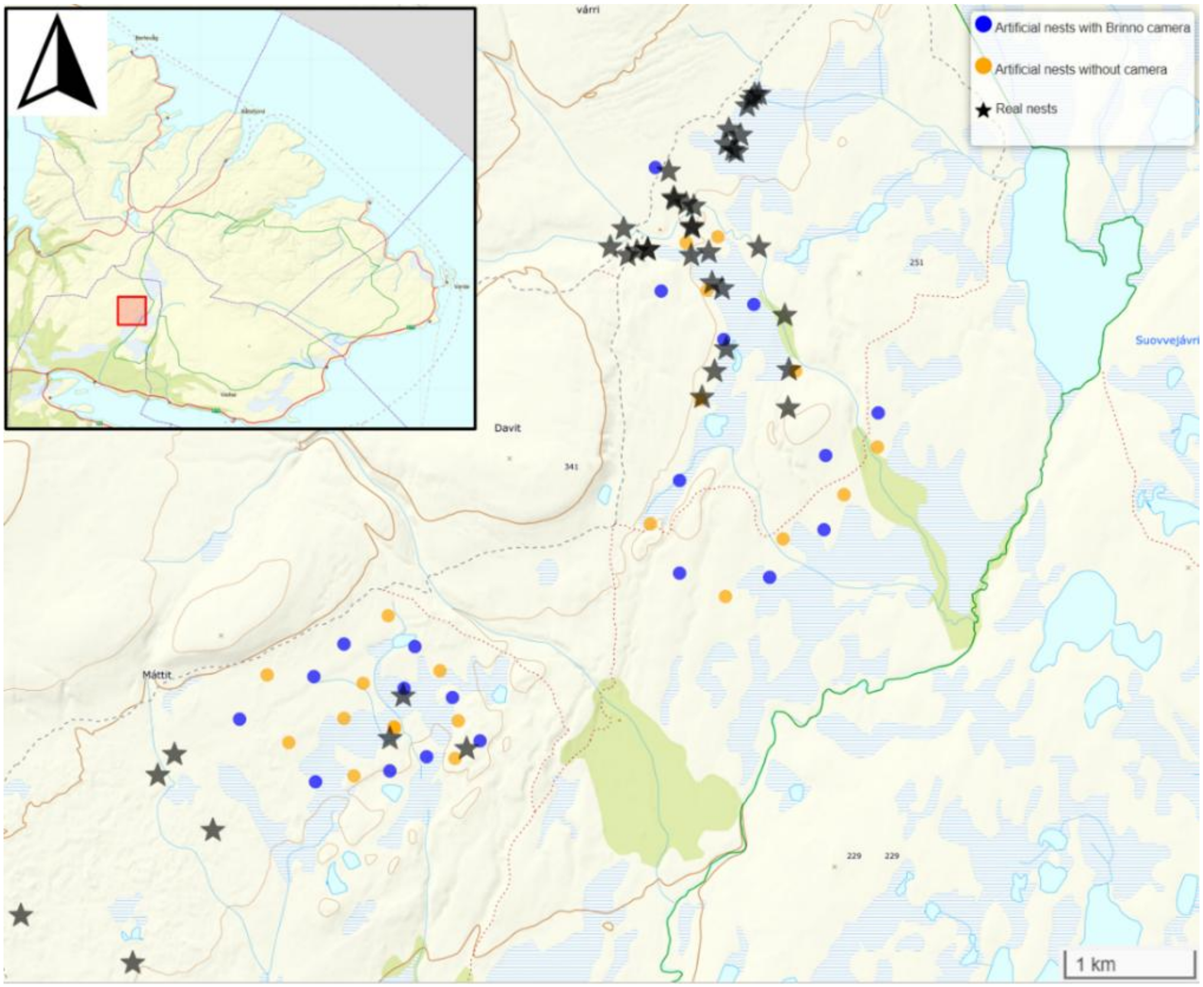


*Figure 1: Location of the research area on the Varanger Peninsula. The 20 artificial nests monitored with a Brinno camera are indicated with a blue circle, the 20 artificial nests without camera monitoring are marked in orange, and the 41 real nests are marked with a star. Basemap from Kartverket (n.d.).*

Additionally, Climate-ecological Observatory for Arctic Tundra (COAT) provided yearly data from 2009 to 2016 on artificial nest predation and rodent trapping at three different locations in the Eastern part of Finnmark: Vestre Jakobselv, Komagdalen, and Ifjordfjellet (Figure 2). Vestre Jakobselv is located close to the study area described above (Figure 1). The regions Vestre Jakobselv and Komagdalen are located on the Varanger Peninsula, whereas the Ifjordfjellet region is situated around the junction of the municipalities of Tana, Gamvik and Lebesby, approximately 100 km west of Komagdalen. The three regions had similar types of vegetational and animal species (Henden et al., 2011; Killengreen et al., 2007). For more detailed information about the study site see Henden et al. (2011) and Ims et al. (2013).

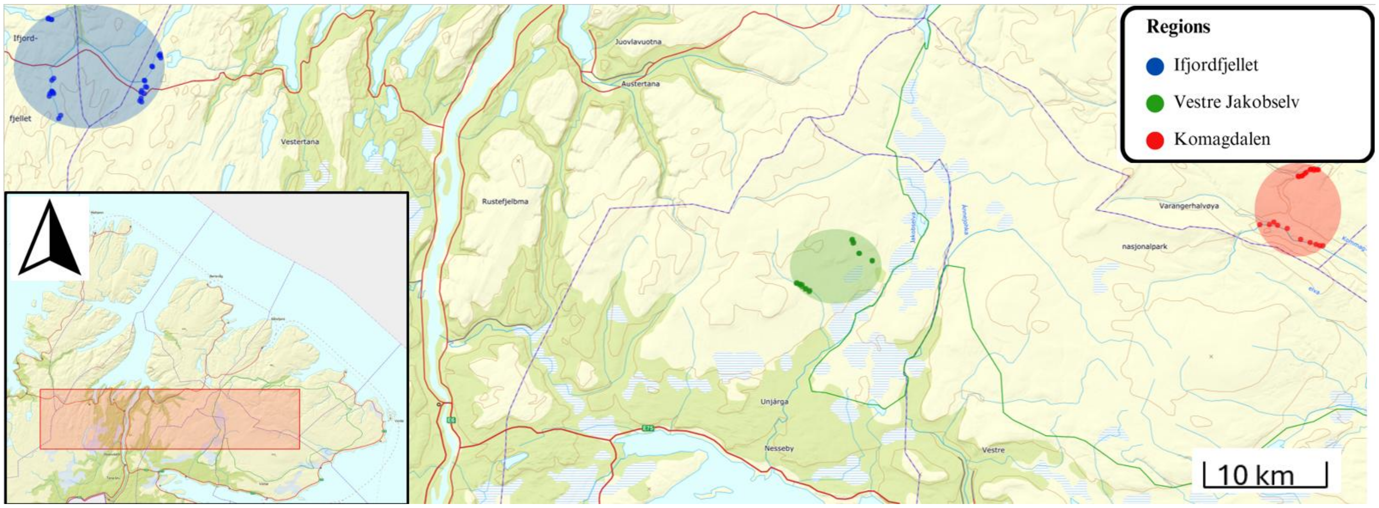


*Figure 2: Location of the rodent traps and the artificial nests in the three different regions; Vestre Jakobselv, Komagdalen and Ifjordfjellet. Each GPS-point indicates a site including both rodent traps and an artificial nest. There were 18 of these sites in Vestre Jakobselv, 22 in Komagdalen and 30 in Ifjordfjellet. Basemap from Kartverket (n.d.).*

## 2.2 Experimental design

This thesis is divided into three distinct sub-studies:

### 1. Evaluating how two types of camera traps affect predation probability on artificial nests: Brinno vs Browning

In this experiment, nest predation probability between artificial nests monitored by a camouflaged and ground-placed Brinno TLC300 (Brinno Inc., Taipei, Taiwan; Figure 3) camera, and artificial nests monitored by Browning Spec Ops Extreme (Browning Trail Cameras, Birmingham, AL, USA) cameras on a pole, were compared. The Brinno camera is smaller and wide-lensed, making it possible to mount the camera close to the nest, as opposed to a conspicuous pole. The data for estimating predation pressure using Brinno cameras were collected during the original fieldwork of this study in 2024. The data for the Browning cameras, however, were obtained from 2018 and 2019 and was provided by Henden et al. (2024). Since the study area is the same in both studies, a direct comparison is possible.

### 2. Does predation probability on artificial nests reflect predation on real nests?

Predation probability on artificial nests were compared with the predation pressure on real nests located in the study area (Figure 1), using data collected during fieldwork in 2024. A total of 41 real nests were located and monitored, as well as separated into two different nest type categories. The nest predation probability on these nest categories was then compared with the predation pressure on 40 artificial nests.

**3. The relationship between rodent abundance and artificial nest predation**

Finally, the relationship between artificial nest predation rates and rodent abundance was analyzed for the regions Vestre Jakobselv, Komagdalen, and Ifjordfjellet (Figure 2). This was made possible by the placement of rodent traps and artificial nests near each other. The data for this analysis was provided by the COAT project.

### *2.2.1 Evaluating how two types of camera traps affect predation probability on artificial nests: Brinno vs Browning*

A total of 40 artificial nests, each consisting of four eggs of common quail, were placed in the study area (Figure 1 and Figure 3). Eggs of common quail are widely used in artificial nest studies (Pärt & Wretenberg, 2002) and were chosen because they resemble those of the native ground-nesting species of interest and are commercially available. The study was carried out from the 23rd of June until the 1st of July in 2024 to match the breeding season of local ground-nesting bird species. The nest locations were randomly selected in suitable habitats for ground-nesting birds, with approximately 500 meters between nests to avoid predation by the same predator individual. A digital map was used to determine the locations, which were then transferred to a GPS device for use in the field. If the exact GPS position was unsuitable for nesting, an alternative location within 15 meters of the original position was selected. Nest IDs ranged from 1 to 40, even-numbered nests were equipped with a Brinno camera, while odd-numbered nests had no camera and served as the control group.

*Figure 3: Artificial nest monitoring setup. The Brinno camera was mounted to two tent plugs with a zip tie, 0.5m from the four quail eggs. To hide the camera from potential nest predators it was covered with camouflage net and tape. A blue tape was used to mark the location. The nests without camera monitoring (control) only consisted of quail eggs and blue tape. Photo: Ekeli, J., 2024.*

At a suitable nest location, the new GPS location was saved, and blue tape was placed on a vegetation branch no more than two meters away from the artificial nest. A small pit was created in the ground using a fist, before adding four quail eggs to the pit. At every other nest, a Brinno camera was mounted about 0.5 meters from the nest (Figure 3). The Brinno TLC300 was selected for its technical specifications, including a wide-angle lens (118 degrees), which allowed for comprehensive (close-up) monitoring of the nest area. Additionally, it has a relatively compact size (dimensions: 6.4 x 5.2 x 10.7 cm) which facilitates discreet placement within the vegetation and minimizes disturbance to the nest environment. To withstand rain and moisture, the cameras were protected by water-proof housing (Brinno ATH120). To fix the cameras towards the artificial nest, they were fastened to two tent pegs with a zip tie. They were then concealed with a camouflage net matching the surroundings, which was secured using a camouflage tape (Figure 3). The time-lapse function was set to capture an image every five seconds, a frequency sufficient to detect predators while also conserving battery life. By testing beforehand, the Brinno was expected to have a Battery-life of seven days with these settings. Images were stored onto SanDisk class 10 SDXC Memory Cards (SanDisk Corp., USA). By not using a camera with motion sensor, red light that could possibly attract predators was avoided. To minimize the risk of detection by predators, the aim was to spend as little time as possible near the nests. The nests and the cameras were collected a week after being placed. If one or more eggs were missing, the nest was considered predated. The predated nests monitored with Brinno cameras were checked for predator confirmation and identification by inspecting the corresponding memory cards.

To compare with the predation probability on the artificial nests monitored by camouflaged Brinno cameras, data from 2018 and 2019 by Henden et al. (2024), was used. Henden et al. (2024) used the larger and more visible (dimensions: 13.7 x 10.6 x 7.5 cm) Browning Spec Ops Extreme cameras fastened on a pole 30-50cm above the ground (Figure 5; Henden et al., 2024). The camera operated with a motion-activated PIR sensor and a time-lapse schedule of one image per hour (Henden et al., 2024). This camera model uses 'no-glow' technology, which causes it to emit infrared light that is invisible to predators in the area during recording (Browning Trail Cameras, n.d.). The 50 nests were in the same area as the Brinno experiment (Figure 1 and Figure 4). Furthermore, 30 of these nests were camera monitored, and 20 nests were not. The methods of the Brinno study were developed from Henden et al. (2024); however, there are several key differences. Firstly, Henden et al.'s research spanned two breeding seasons (2018 and 2019). In addition, the Browning study examined the effect of vegetation height on predation rates (which was found), whereas in the Brinno study, the nests were placed in areas with similar vegetational height. Lastly, Henden et al. (2024) alternated between using four and eight quail eggs in their artificial nests, though no significant effect was found of this variation. Consequently, this factor was not incorporated into the Brinno study and will not be considered when comparing results. The most important difference between the two studies being the monitoring time, as Henden et al. (2024) monitored their nests for 14 days, compared to seven days in the Brinno experiment. The longer exposure time needs to be accounted for statistically. The methodology for the Browning study is fully described in Henden et al. (2024).

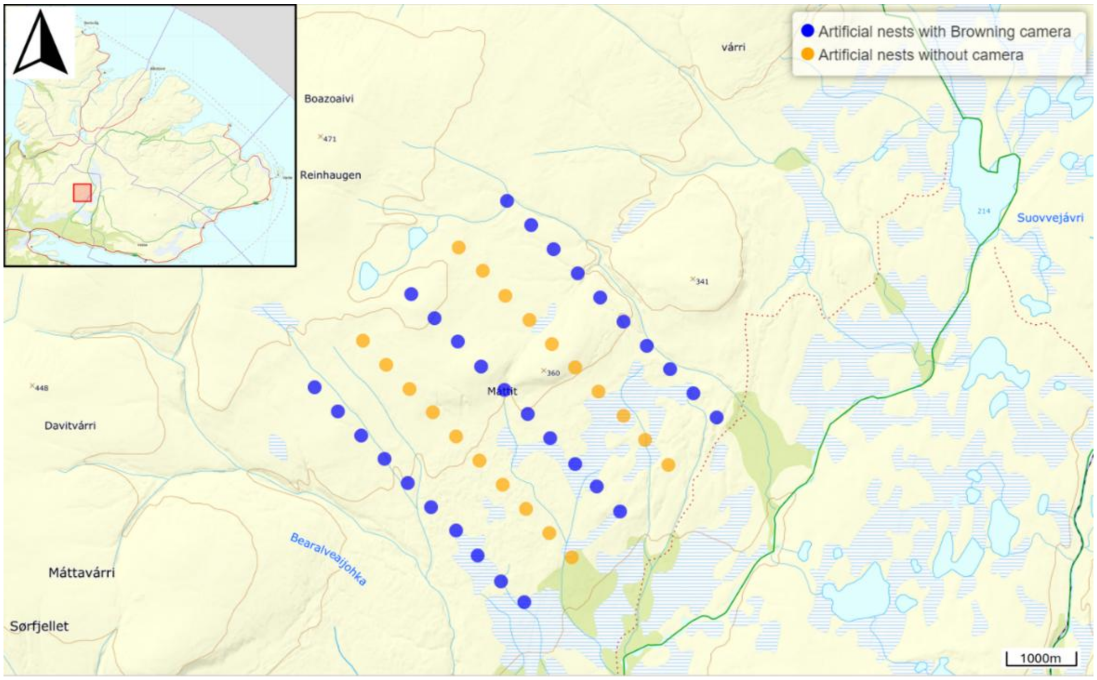


*Figure 4: Location of the Henden et al. (2024) experiment, in the same area as the artificial and real nests in Figure 1. The 30 artificial nests with Browning camera monitoring are marked in blue, and the 20 artificial nests without camera monitoring are marked in orange. Basemap from Kartverket (n.d.).*

*Figure 5: Camera monitoring method used in Henden et al. (2024). Browning trail cameras were fastened to a 30–50 cm pole. The note was used only for photographic nest identification. Photo: Strømeng, M. A., 2018. Photo used with permission (pers. comm. from Henden et al., 2024).*

### *2.2.2 Does predation probability on artificial nests reflect predation on real nests?*

To compare with the predation probability of the artificial nests from the Brinno research, 41 real nests of local ground-nesting bird species were located (Figure 1 and Appendix E). The species of nests located were grouped into two categories due to differences in nest type. Passerines which included meadow pipit, Lapland longspur and bluethroat (*Luscinia svecica*), and non-passerines including willow ptarmigan, wood sandpiper (*Tringa glareola*), common snipe (*Gallinago gallinago*), dunlin, greater scaup (*Aythya marila*), Eurasian Golden Plover, ruff (*Calidris pugnax*), and common redshank (*Tringa totanus*). The nests of the willow ptarmigan and shorebirds were similar but differed to the smaller and more well-hidden passerine nests. The passerine nests were typically hidden within a tuft, whereas the non-passerines had their nests placed straight on the ground, primarily camouflaged by the cryptic appearance of the incubating bird.

The location was done by a combination of using expert nest locaters, paying attention to alarm calls and behavior, and the use of 'rope-dragging' to flush the incubating bird from the nest location (see Appendix B, Labisky, 1957; Machín et al., 2019). When a nest was located, the GPS coordinates were recorded, and a blue marker was placed on a vegetational branch within two meters of the nest. Additionally, real nests were discovered while placing out the artificial ones, indicating that the artificial nests were placed in suitable habitat. Each nest was photographed, and the bird species and the number of eggs or chicks were documented. The nests were then monitored by checking them every two to four days, for a maximum of nine days. For the real nests the absence of eggs could mean hatching. This was particularly difficult with the shorebirds (non-passerine group) as they are precocial and are known to remove eggshells from their nests (Tinbergen et al., 1962). As a result, five nests had an uncertain fate. To address this uncertainty, predation events were categorized as either 'certain' or 'possible'. As with the installation of the artificial nests, time spent at nest sites was minimized to avoid detection by predators and reduce disturbance to the birds. Exercising caution in this step is even more important with real nests, as this is a vulnerable life stage for the birds.

### *2.2.3 The relationship between rodent abundance and artificial nest predation*

To investigate the relationship between rodent abundance and nest predation, yearly data provided by COAT from 2009 to 2016 were analyzed. Both 12 snap traps for small rodents and an artificial nest were located at each site (a 15 x 15 m grid; Figure 2). The artificial nests were made by creating a small pit in the ground with a fist. The nests contained two quail eggs, and if one of the eggs was missing the nest was considered predated. Given the proximity of the nest to the rodent snap traps, the comparison between the two should be highly relevant. A total of 18 sites were set up in Vestre Jakobselv, 22 in Komagdalen and 30 in Ifjordfjellet. Each quadrant included 12 snap traps, with three traps placed in a radius of two meters around each corner of the quadrant (Myllymäki et al., 1971). The snap traps were operated for two trapping days in the beginning of the summer (early July) and two in the autumn (early September), each year (Henden et al., 2011). The nests were checked every five days over a total monitoring period of 15 days (Ims et al., 2013). The methodology used in the collection of this data is described fully in Ims et al. (2013). Study area and experimental design is partly based on the term paper from the preparatory master’s course 'Methods in Natural Science' (Ekeli, 2024).

## 2.3 Statistical analyses

### *2.3.1 Evaluating how two types of camera traps affect predation probability on artificial nests: Brinno vs Browning*

In this test the effects of camera type and exposure time on predation probability were examined (Figure 1 and Figure 2). The different categories in camera type included Brinno, Browning and none (control). As the artificial nests with Browning cameras had a longer exposure time, 14 days compared to seven days in our study with Brinno, this difference needed to be accounted for in the analysis. The predation rate for each camera type category was calculated by dividing the total number of predation events by the total monitoring time (in days) for nests within each category. Predation is binary, meaning a nest can either be predated or not. To analyze this, generalized linear models (GLM) with a binomial family were used to model predation probability as a function of camera type and monitoring time.

In the first model, hereby: the 'offset model', an offset was included to adjust for the effect of exposure time on predation probability. Specifically, the number of days the different nest categories were monitored was included as a log-transformed offset in the model. This was done to ensure comparability of predation probability across the different camera types (Brinno, Browning and control). The offset model can be expressed mathematically as follows:

$$\boldsymbol{log\left(\frac{\pi}{1-\pi}\right) = \beta_0 + \beta_1 \cdot CameraTypeBrowning + \beta_2 \cdot CameraTypeNone + log(DaysMonitored)} \tag{1}$$

In this logistic regression model, $\pi$ represents the probability of a predation event occurring. The model estimates the log-odds of predation, with $\beta_0$ serving as the intercept. This reflects the log-odds of nest predation when Brinno is used for monitoring the artificial nests. The coefficient $\beta_1$, however, represents the change in the log-odds of predation when using a Browning camera, relative to the Brinno camera. Similarly, $\beta_2$ captures the change in the log-odds of predation when no camera is present, still relative to the Brinno camera. Additionally, the term log(DaysMonitored) is included as an offset in the model. This offset adjusts for the difference in exposure time for the Browning and Brinno cameras.

In the second model included in this test, hereby: the 'predictor model', the effect of exposure time on predation was examined by including the number of days monitored per nest as a predictor in the generalized linear model. The predictor model can be expressed by the following formula:

$$\boldsymbol{log\left(\frac{\pi}{1-\pi}\right) = \beta_0 + \beta_1 \cdot CameraTypeBrowning + \beta_2 \cdot CameraTypeNone + \beta_3 \cdot DaysMonitored} \tag{2}$$

The variables are the same as in the offset model, except for $\beta_3$, which represents the effect of exposure time. For each additional day the artificial nest is monitored the log-odds of predation increases with $\beta_3$. To visualize the effect of exposure time on predation, in the predictor model, the 'ggpredict' function from the 'ggeffects' package (Lüdecke, 2018) was used. This function allowed for the visualization of predicted predation probability at the given artificial nests for different camera types with 7, 12 and 22 days of exposure time. The incubation period for the different passerine and non-passerine species in the study area is approximately 12 and 22 days, respectively (Evans et al., 2005; Westerskov, 1956). The Brinno study lasted seven days, while the predictor model allows for the visualization of the predation probability for the Browning method had it also been conducted for seven days.

### *2.3.2 Does predation probability on artificial nests reflect predation on real nests?*

In this test, the difference in nest predation probability based on nest category was examined (Figure 1). The three different nest categories were artificial, passerine and non-passerine. The predation rate was calculated by dividing the number of nests for each category by the number of predation events. Because of uncertainty regarding the fates of some of the real nests in the non-passerine category, two GLMs were used: one for the upper estimate of nest predation incidents and one for the lower estimate. The response variable was binary predation.

$$log\left(\frac{\pi}{1-\pi}\right) = \beta_0 + \beta_1 \cdot Category_{Non-passerine} + \beta_2 \cdot Category_{Passerine} \quad (3)$$

In this formula, $\beta_0$ represents the log-odds of predation for the intercept, which is the artificial nest category. Furthermore, $\beta_1$ represents the range of log-odds of nest predation for the non-passerine category, with lower and upper estimates highlighting the uncertainty in predation for this category. Finally, $\beta_2$ represents the effect of the passerine category on the log-odds of nest predation. To assess the overall significance of nest category in predicting predation probability, an analysis of deviance using a likelihood ratio test was performed. Estimated marginal means (emmeans) were derived from the fitted GLMs to summarize the predicted nest predation risk for each nest category. These means were then compared using post-hoc pairwise tests with a Tukey adjustment to evaluate whether the differences between categories were statistically significant.

### *2.3.3 The relationship between rodent abundance and artificial nest predation*

In this test, the correlation between predation rates on artificial nests and the estimated number of rodents from a time series spanning 2009 to 2016, was studied. This analysis was conducted for the regions Ifjordfjellet, Komagdalen and Vestre Jakobselv (Figure 3). The nest predation rate was determined by the amount of predation incidents divided by the number of nests per year in each region. There were 18 nests in Vestre Jakobselv, 30-32 (difference between the years) in Komagdalen and 26-30 in Ifjordfjellet. Rodent trapping effort was determined per year for the three different locations. The calculation was based on the number of snap traps per quadrant (12), the number of quadrants per location and the number of days operative per year (four). This gave an effort of 864 trapping days per year for Ifjordfjellet, 648 trapping days for Vestre Jakobselv and 888 trapping days for Komagdalen. To adjust for these differences the number of rodents trapped at each location for each year in the time series was converted into a 'rodent index'. The rodent index represents the number of rodents trapped each year per 100 trapping days (Nelson & Clark, 1973). To account for potential delayed effects of rodent abundance on nest predation rates, a variable called 'lagged rodent index' was created. The delayed effect is expected due to predators' numerical response to the increased abundance of rodents (Gilg et al., 2003). The lagged index aligns the rodent index from one year (n), with the nest predation rate from the year after (n+1). In addition, potential immediate effects from rodent index on artificial nest predation rate. The generalized linear model tested the binary response variable predation with the fixed effects of rodent index and lagged rodent index, with location as a random effect. This model can be expressed with the following formula:

$$\boldsymbol{log\left(\frac{\pi}{1-\pi}\right) = \beta_0 + \beta_1 \cdot Rodent\ index + \beta_2 \cdot Lagged\ Rodent\ index + u_{region}} \qquad \textbf{(4)}$$

In this formula, $\beta_0$ represents the intercept of the model, which corresponds to the log-odds of predation when the rodent index and the lagged rodent index are both 0. The $\beta_1$ variable, however, reflects the effect of the current rodent index on the log-odds of predation, while $\beta_2$ represents the effect of the lagged rodent index on the log-odds of predation. Lastly, the formula expresses the models' random effect for each region, represented by $u_{region}$, which accounts for variability in predation rates across different the three different regions.

Finally, a test was also conducted to assess whether the 'lagged rodent index – nest predation rate effect' differed between the three regions. Three GLMs with a binomial response variable were fitted using a logit link function. The models were applied to the different regions: Ifjordfjellet, Komagdalen, Vestre Jakobselv, using a subset of the dataset. For each region-specific model, the response variable was predation, and the predictor variables included the lagged rodent index and the present rodent index. The log-odds of artificial nest predation were modeled as a function of rodent abundance using the following equation:

$$logit(P(Predation)) = \beta_0 + \beta_1 \cdot Lagged\ Rodent\ index + \beta_2 \cdot Rodent\ index \qquad (5)$$

Even though an overall significant positive relationship between the lagged rodent index and the nest predation rate was found (Table 5), it was suspected that this was more applicable to some regions than others (Figure 12). Therefore, the relationship between the lagged rodent index and nest predation was tested for each region (Ifjordfjellet, Vestre Jakobselv and Komagdalen). In this equation (5), $\beta_0$ represents the intercept, which corresponds to the baseline log-odds of predation when both the lagged rodent index and rodent index are zero. $\beta_1$ captures the effect of lagged rodent index, while $\beta_2$ represents the effect of the current rodent index.

All models from these sub studies were fitted using the 'glmer' function from the 'lme4' package (Bates et al., 2015) in RStudio version 2024.04.2, Build 764 (RStudio Team, 2024). This version of RStudio was also used for the rest of the programming in this study. Additionally, 'tidyverse' and 'dplyr' were used for data manipulation, and 'ggplot2' for data visualization (Wickham, 2011; Wickham et al., 2019; Wickham et al., 2023). Furthermore, the maps (Figure 1, Figure 2 and Figure 4) were made by the usage of the 'leaflet' package (Cheng et al., 2019). The equations were extracted from the models using the 'coef' function from the 'stat' package (Chambers & Hastie, 1992). Lastly, this study utilized ChatGPT (Version 4, November 2024 to March 2025) for assistance in interpreting the output from statistical models, troubleshooting code for maps and plots, and correcting grammar.

# 3. Results

## 3.1 Evaluating how two types of camera traps affect predation probability on artificial nests: Brinno vs Browning

Out of 40 artificial nests, a total of seven were predated. Three of the predated nests were monitored by a camouflaged Brinno camera, while the other four had no camera. Also, at one Brinno monitored nest a long-tailed skua (*Stercorarius longicaudus*) moved an egg just inches away from the nest without consuming it (Appendix D). Of the three predated nests with cameras, only two predation incidents were recorded, as one of the camera batteries had been depleted. In the two cases where identification was possible, the predators were the long-tailed skua (Figure 7) and the Arctic skua (*Stercorarius parasiticus*; Appendix C), showing that there was a high proportion of skuas among the nest predators (Figure 6). Out of the 20 Brinno cameras that were used in this study, two cameras had empty batteries before the seventh day. One camera had a full battery after seven days (could be due to human error), and the rest of the cameras were low in battery. The cameras captured predation events clearly (Figure 7 and Appendix C). They also seemed to handle heavy rain and wind effectively, as the weather varied a lot during the study period.

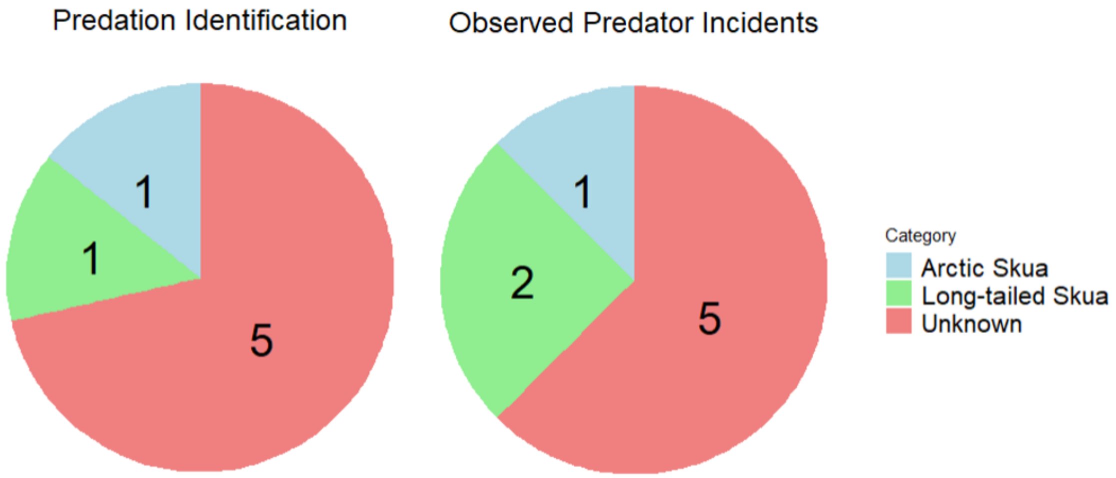


*Figure 6: Left – Proportion of nest predator species identified and associated with a confirmed predation event. Right – Proportion of observed predator species showing interest in the nest, including both confirmed nest predators and those not observed preying.*

*Figure 7: Predator identification, a long-tailed skua consuming quail eggs from a Brinno monitored artificial nest. Still image from the surveillance footage.*

When comparing results with Henden et al. (2024)'s research with the use of Browning trail cameras, the camera types had significantly different nest predation probability. This pattern remained even when taking the monitoring duration into consideration, as in the offset model (Residual deviance = 121.42 on 136 degrees of freedom, AIC = 127.42, n = 139; Table 1 and Figure 8). Including exposure time as a predictor did not improve the model fit in the predictor model either (Residual deviance = 120.36 on 135 degrees of freedom, AIC = 128.36, n = 139; Table 2 and Figure 9).

Predation probability was significantly higher for nests monitored with Browning cameras in the offset model, with an estimated increase in log-odds of predation of 3.68 (SE = 0.81, $p < 0.001$, n = 139; Table 1) compared to the reference group (Brinno camera), corresponding to approximately 39 times higher odds of predation. In contrast, nests with no cameras did not differ significantly from the reference group (p = 0.23), indicating that Brinno cameras did not influence nest predation risk.

*Table 1: Summary of coefficients in the offset model, a generalized linear model (GLM) testing camera type significance on nest predation probability. This model used offset to account for the difference in exposure time for the Browning and Brinno monitored artificial nests.*

| Predictor | Estimate | SE | z-value | p-value |
|---|---|---|---|---|
| **Intercept (Brinno)** | -3.68 | 0.63 | -5.88 | <0.001 |
| **Browning** | 3.68 | 0.81 | 4.53 | <0.001 |
| **None** | 0.81 | 0.68 | 1.20 | 0.23 |

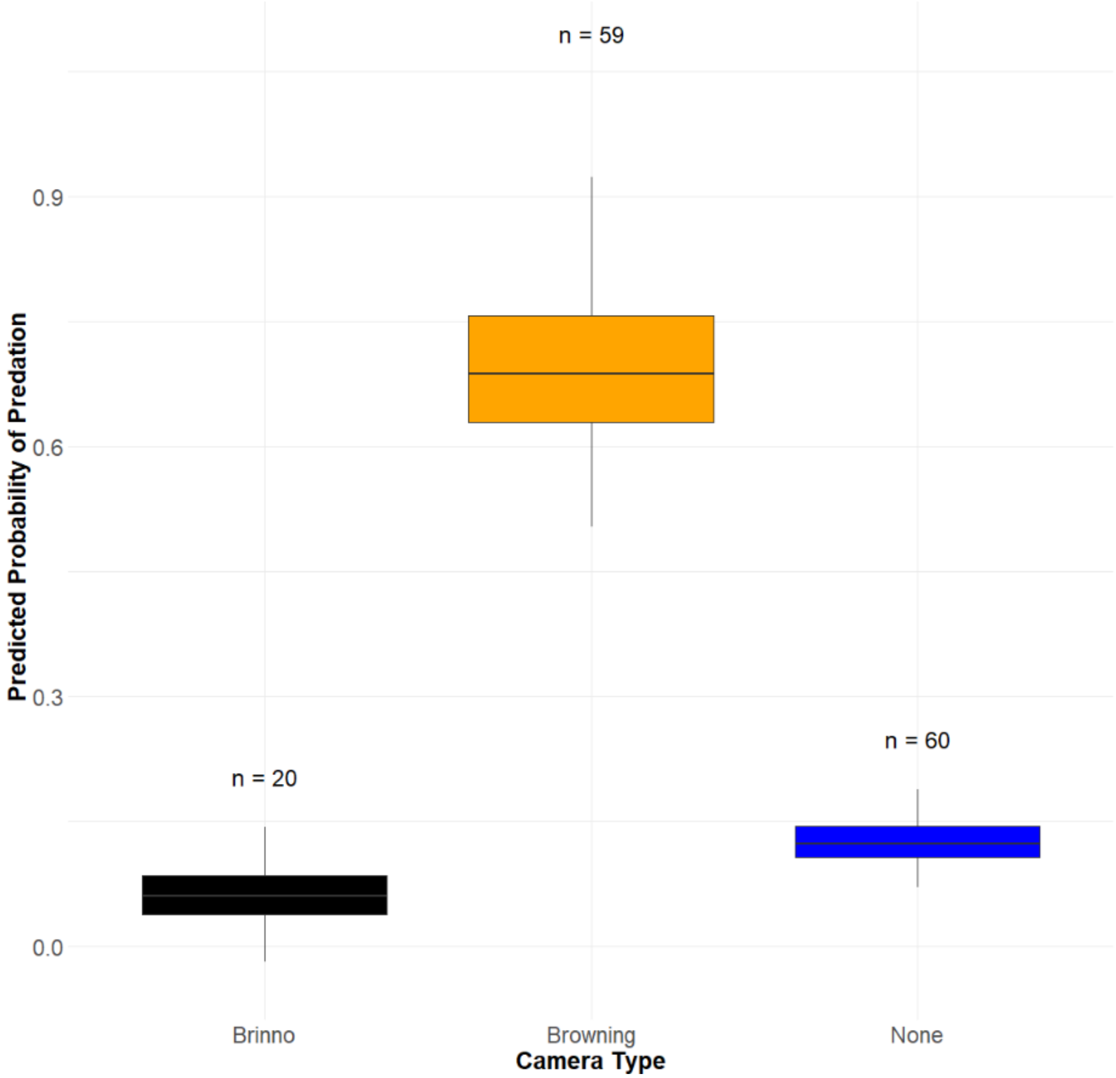


*Figure 8: Predicted probability of predation for artificial nests monitored with a Brinno camera, Browning camera, or no camera, based on a model using logarithmic offset to account for the difference in monitoring time for the different camera types.*

Further testing showed that the duration of monitoring time had a statistically significant effect on the predation probability, with an estimate of 0.19 (SE = 0.09, z = 2.07, p = 0.038; Table 2). This indicated that the likelihood of predation increased by 21% for each additional day of monitoring, on average across all categories. Once again, the predation probability of nests with no camera did not significantly differ from Brinno-monitored nests (Table 2). The predictor model illustrated how predation probability would vary with predictive monitoring times of 7, 12 and 22 days (Figure 9). This prediction showed that the differences in nest predation probability effect between Brinno and Browning became smaller with an increased exposure time.

*Table 2: Summary of model coefficients for the predictor model, testing both the effects of camera type and exposure time on predation probability.*

| Predictor | Estimate | SE | z-value | p-value |
|---|---|---|---|---|
| **Intercept** | -3.07 | 0.90 | -3.41 | <0.001 |
| **Browning** | 3.04 | 1.04 | 2.93 | 0.003 |
| **None** | 0.35 | 0.84 | 0.42 | 0.68 |
| **Days monitored** | 0.19 | 0.09 | 2.07 | 0.04 |

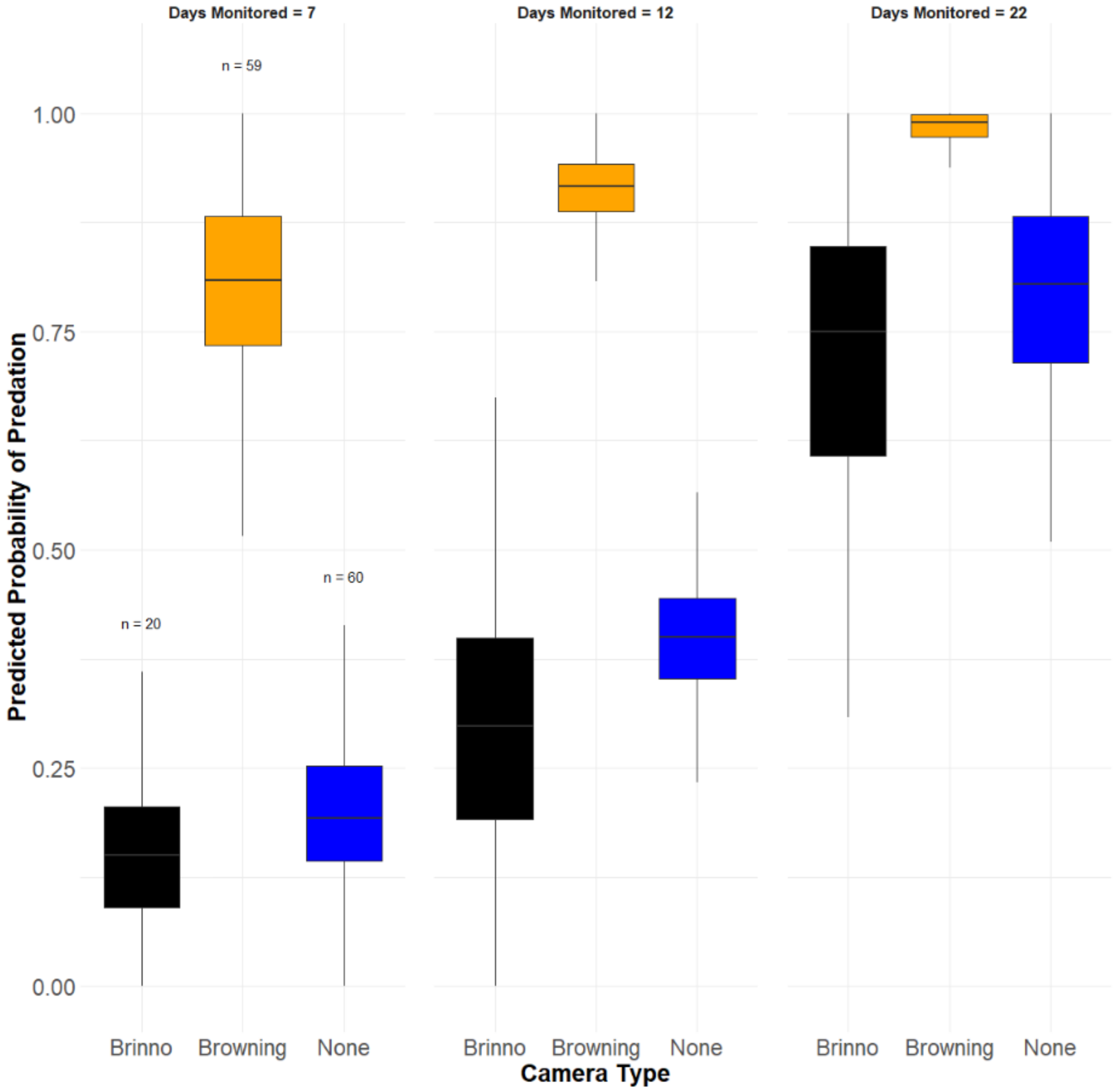


*Figure 9: Predicted probability of predation on artificial nests monitored with Brinno, Browning, and no camera, based on the predictor model. These predictions are based on simulated monitoring durations of 7, 12, and 22 days. The local passerine birds typically incubate for approximately 12 days, while the non-passerines have an incubation time of approximately 22 days. In total there were 20 nests monitored with Brinno cameras (2024), 59 nests monitored with Browning (2018 and 2019) and 60 nests with no camera (2018, 2019 and 2024).*

## 3.2 Does predation probability on artificial nests reflect predation on real nests?

For the real nests, there was one certain predation event on a willow ptarmigan nest and five uncertain predation events on other non-passerine nests (Figure 10 and Appendix E). Thus, an upper estimate for non-passerine predation events was calculated by including both certain and uncertain events, while the lower estimate included only the one confirmed predation event. For the 19 passerine nests monitored, no predation events were observed (Appendix E). These nests were smaller and more well-hidden than the non-passerine and artificial nests (Figure 10 and Figure 11). To compare these findings, artificial nest predation data from fieldwork in 2024 was used. Because no significant difference was found in predation probability between the artificial nests monitored with Brinno camera and the nests without a camera, these results were combined into one category: artificial nests.

*Figure 10: Left - A willow ptarmigan (non-passerine) nest with certain signs of predation. Right - The same nest four days prior to the observation of the predation event. Photos: Ekeli, J., 2024.*

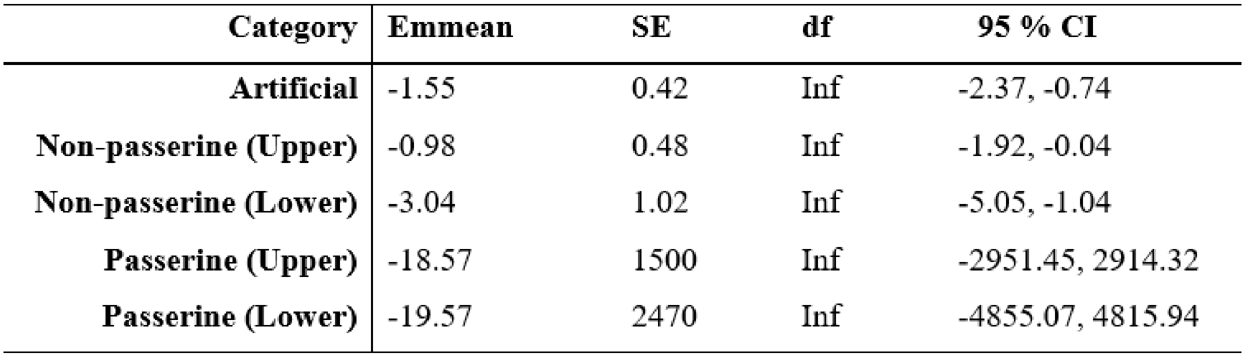
*Figure 11: A typical passerine nest - small, well-hidden within a tuft and under a branch. This is a meadow pipit nest, the most frequently encountered of this type. Photo: Ekeli, J., 2024.*

When comparing the probability of predation between artificial and real nests, a likelihood ratio test revealed a significant overall effect of nest category on the probability of predation (Residual deviance = 45.23 on 78 degrees of freedom, AIC = 51.23, p = 0.03, n = 81; Appendix A). This indicates that nest type significantly influenced predation risk.

Although the overall effect of nest category on the probability of predation was statistically significant, post hoc comparisons between individual groups were not. This suggests that while nest type influences predation risk, the differences between artificial, passerine, and non-passerine nests were not significant when compared pairwise (Table 3). Predicted log-odds were similar, especially when taking into consideration confidence intervals, standard error, and upper and lower estimates of predation (Figure 12).

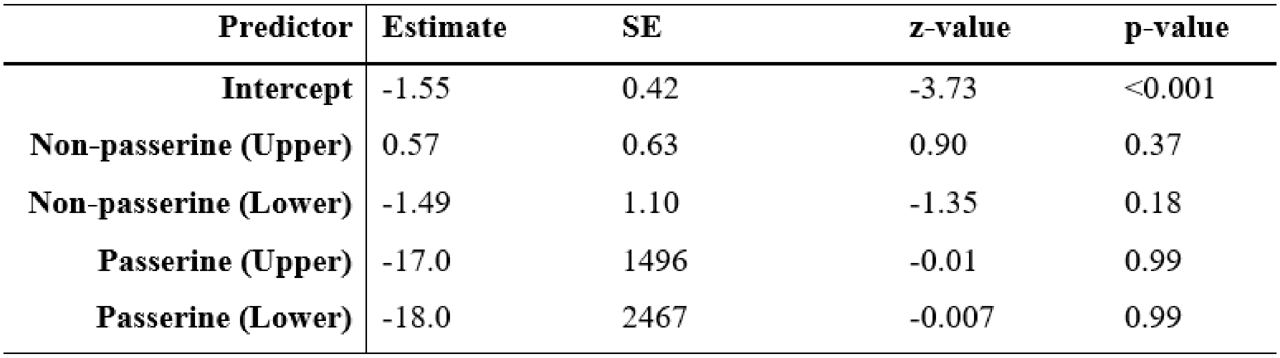
For further testing, the lower estimate of predation probability for the real nests was used, as it provided the most conservative basis for detecting a significant difference from the seven predation events observed in the artificial nests. The generalized linear model (GLM) did not yield statistically significant results (Residual deviance = 45.234 on 78 degrees of freedom, AIC = 51.234, n = 81; Table 4). To assess whether the absence of observed predation in passerine nests influenced the model outcomes, the passerine data were excluded and the analyses re-run. However, this exclusion also failed to produce significant results.

*Table 3: Post-hoc pairwise comparison of nest categories. The different means suggested different nest predation probability across the categories. The emmean of the passerine group varied depending on whether the lower or upper estimate of non-passerine was used, hence the different passerine categories.*



| **Category** | **Emmean** | **SE** | **df** | **95 % CI** |
|---:|---|---|---|---|
| **Artificial** | -1.55 | 0.42 | Inf | -2.37, -0.74 |
| **Non-passerine (Upper)** | -0.98 | 0.48 | Inf | -1.92, -0.04 |
| **Non-passerine (Lower)** | -3.04 | 1.02 | Inf | -5.05, -1.04 |
| **Passerine (Upper)** | -18.57 | 1500 | Inf | -2951.45, 2914.32 |
| **Passerine (Lower)** | -19.57 | 2470 | Inf | -4855.07, 4815.94 |

*Table 4: Summary of model coefficients for the test comparing artificial nest predation probability and the upper and lower estimate of real nest predation probability.*



| **Predictor** | **Estimate** | **SE** | **z-value** | **p-value** |
|---:|---|---|---|---|
| **Intercept** | -1.55 | 0.42 | -3.73 | <0.001 |
| **Non-passerine (Upper)** | 0.57 | 0.63 | 0.90 | 0.37 |
| **Non-passerine (Lower)** | -1.49 | 1.10 | -1.35 | 0.18 |
| **Passerine (Upper)** | -17.0 | 1496 | -0.01 | 0.99 |
| **Passerine (Lower)** | -18.0 | 2467 | -0.007 | 0.99 |

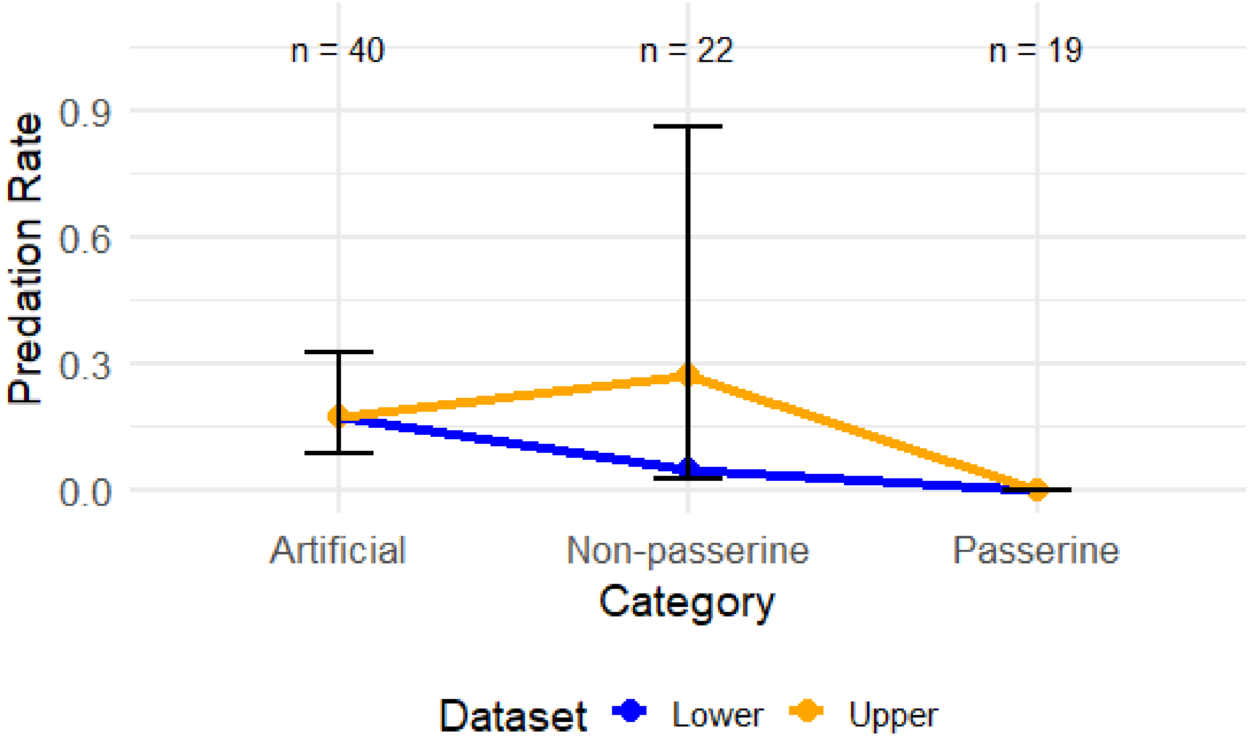


*Figure 12: Predicted log-odds of predation for artificial nests, upper and lower estimates for real non-passerine nests, and passerine nests. The non-passerine nest category has a lower and upper estimate of predation incidents, as there were nests with uncertain faiths. The CI (confidence interval) for passerines is 0 due to 0 predation events for this category. The large CI for the non-passerine category (CI for both lower and upper estimate for this category combined) is due to the large uncertainty as well as the small sample size.*

### 3.3 The relationship between rodent abundance and artificial nest predation

The generalized linear model (GLM) that tested the correlation between rodent abundance and artificial nest predation (AIC = 770.2, deviance = 762.2 on 628 df.resid, n= 632) indicated a significant effect of the lagged rodent index on artificial nest predation rate (Estimate = 0.06, SE = 0.03, z = 2.23, p = 0.03; Table 5). This suggested that an increased rodent abundance for year n was associated with a higher likelihood of predation on artificial nests in year n+1 (Figure 13). However, the rodent index for the same year as the nest predation failed to show significant results (Estimate = -0.02, SE = 0.02, z = -0.83, p = 0.41).

*Table 5: Summary of model coefficients for the generalized linear model (GLM) testing lagged rodent index effect of artificial nest predation. There were 235 artificial nests and 864 trapping nights in Ifjordfjellet, 253 nests and 888 trapping nights in Komagdalen and 144 nests and 648 trapping nights in Vestre Jakobselv.*

| Predictor | Estimate | SE | z-value | p-value |
|---|---|---|---|---|
| **Intercept** | -0.16 | 0.56 | -0.29 | 0.78 |
| **Rodent index** | -0.02 | 0.02 | -0.83 | 0.41 |
| **Lagged rodent index** | 0.06 | 0.03 | 2.24 | 0.03 |

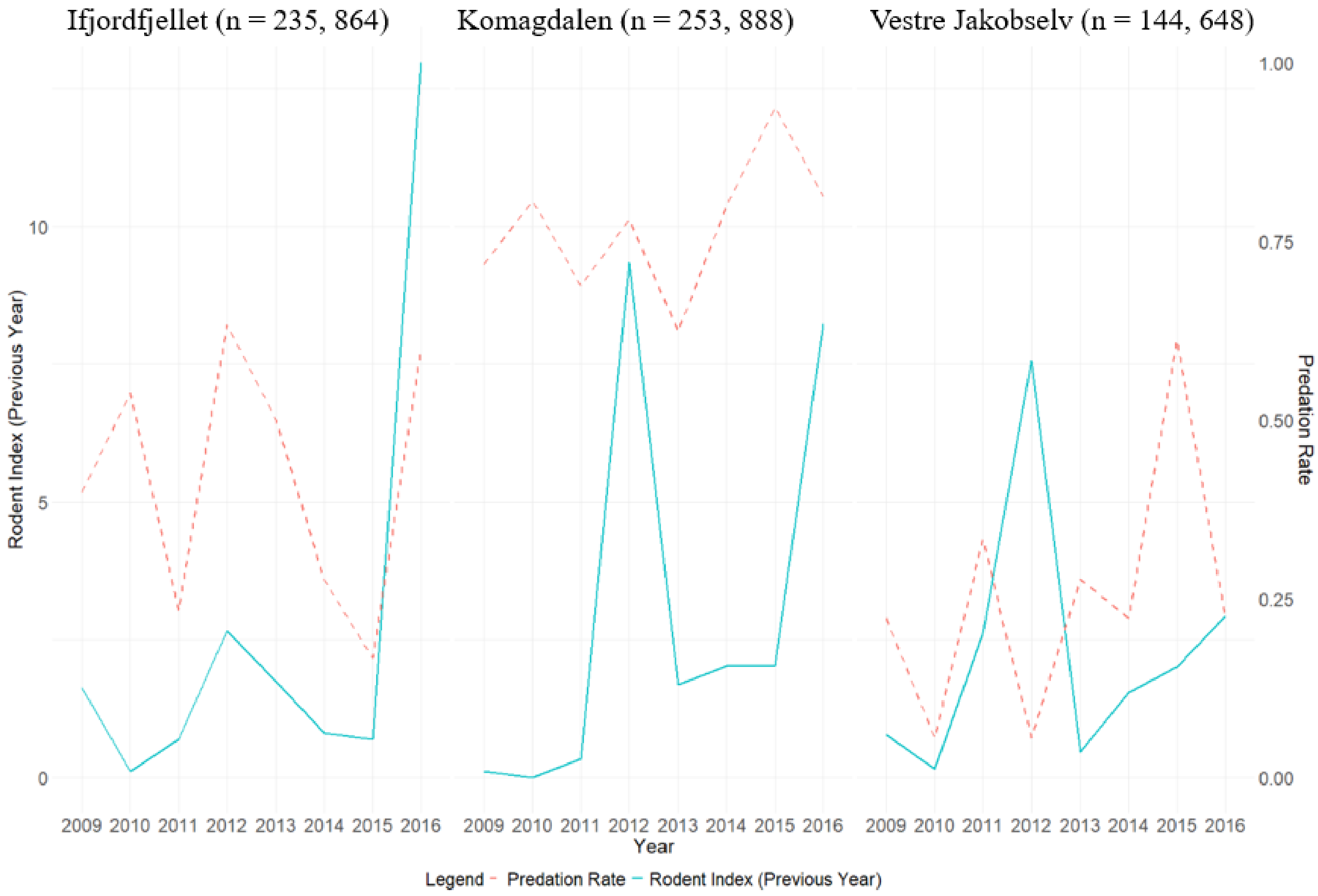
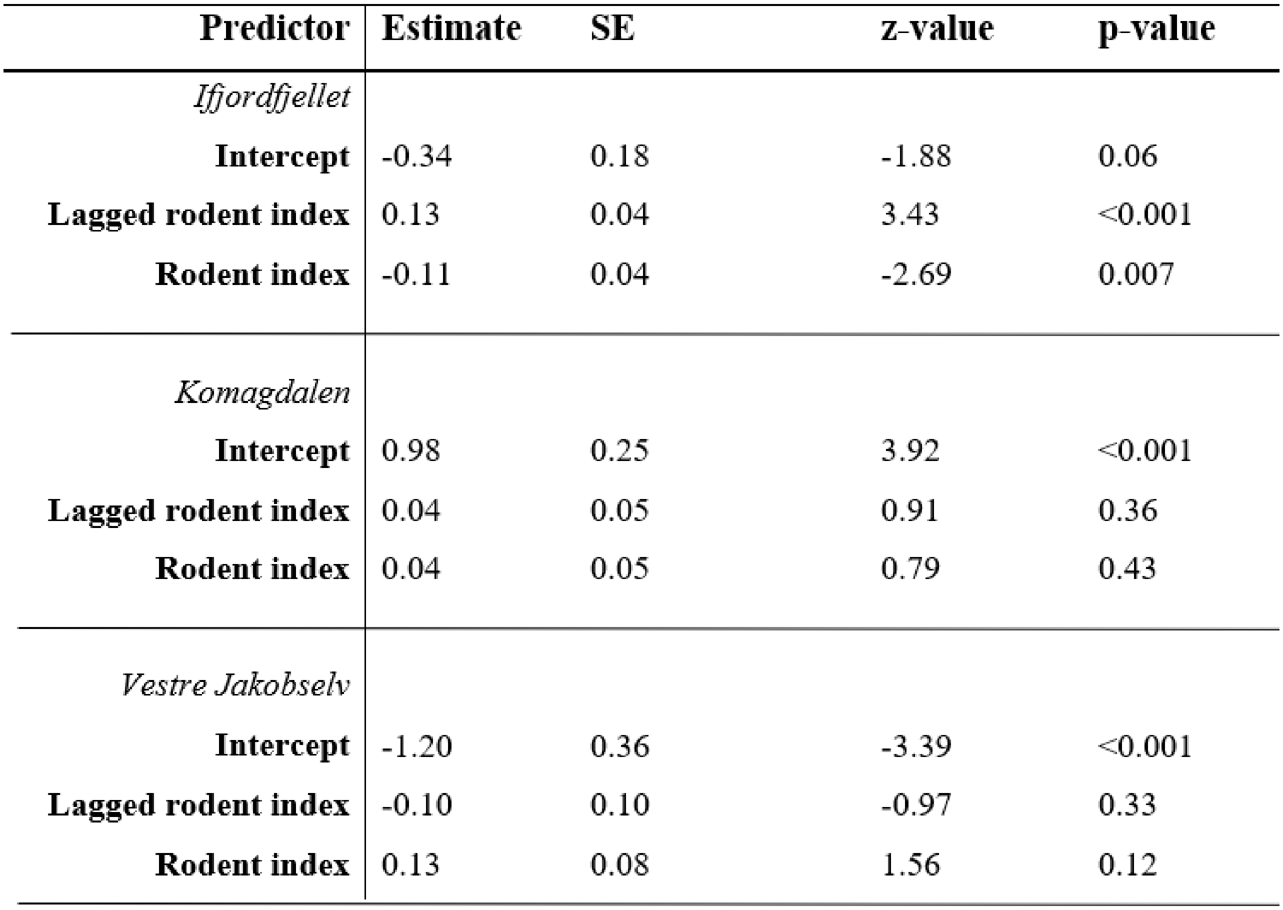


*Figure 13: Relationship between the artificial nest predation rate and the rodent index from the previous year in Ifjordfjellet, Komagdalen and Vestre Jakobselv. The rodent index was calculated by dividing the number of rodents trapped per region each year by the number of trapping nights (n), then multiplying by 100 to represent the number of rodents per 100 trapping nights. The nest predation rate was calculated by dividing the number of predation incidents by the total number of nests (n) per year at each location.*

*Table 6: Summary of GLMs coefficients from the test assessing the relationship between both lagged and present rodent index and artificial nest predation across the three regions Ifjordfjellet, Komagdalen and Vestre Jakobselv. A significant positive relationship between lagged rodent index and nest predation was only found in Ifjordfjellet.*



| **Predictor** | **Estimate** | **SE** | **z-value** | **p-value** |
|---|---|---|---|---|
| *Ifjordfjellet* | | | | |
| **Intercept** | -0.34 | 0.18 | -1.88 | 0.06 |
| **Lagged rodent index** | 0.13 | 0.04 | 3.43 | <0.001 |
| **Rodent index** | -0.11 | 0.04 | -2.69 | 0.007 |
| *Komagdalen* | | | | |
| **Intercept** | 0.98 | 0.25 | 3.92 | <0.001 |
| **Lagged rodent index** | 0.04 | 0.05 | 0.91 | 0.36 |
| **Rodent index** | 0.04 | 0.05 | 0.79 | 0.43 |
| *Vestre Jakobselv* | | | | |
| **Intercept** | -1.20 | 0.36 | -3.39 | <0.001 |
| **Lagged rodent index** | -0.10 | 0.10 | -0.97 | 0.33 |
| **Rodent index** | 0.13 | 0.08 | 1.56 | 0.12 |

The GLMs assessing the relationship between rodent abundance and artificial nest predation showed varying effects across regions (Table 6). For Ifjordfjellet, the model (AIC = 309.41, residual deviance = 303.41 on 232 df, n = 235) revealed a significant positive effect of the lagged rodent index on nest predation (Estimate = 0.13, SE = 0.04, z = 3.43, $p < 0.001$; Table 6). This indicated that higher rodent abundance in the previous year was associated with an increased likelihood of nest predation. In contrast, the rodent index for the current year showed a significant negative effect on predation (Estimate = -0.11, SE = 0.04, z = -2.69, p = 0.007; Table 6.).

For Komagdalen, however, the model (AIC = 277.18, residual deviance = 271.18 on 250 df, n = 253) did not detect a significant effect of either rodent index on predation (Lagged Rodent Index: Estimate = 0.04, SE = 0.05, z = 0.91, p = 0.36; Rodent Index: Estimate = 0.04, SE = 0.05, z = 0.79, p = 0.43; Table 6.).

Similarly, in Vestre Jakobselv, the model (AIC = 164.37, residual deviance = 158.37 on 141 df, n = 144) found no significant effects of rodent abundance on predation (Lagged Rodent Index: Estimate = -0.10, SE = 0.10, z = -0.97, p = 0.33; Rodent Index: Estimate = 0.13, SE = 0.08, z = 1.56, p = 0.12; Table 6.).

# 4. Discussion

The results indicate that the monitoring method with Brinno affected the predation probability of artificial nests less than the commercially available Browning trail camera. Predation probability on artificial nests monitored with Brinno were not significantly different from the artificial nests in the control group, which lacked camera monitoring, indicating that this method did not affect predation pressure. Furthermore, there was an overall significant difference in the predation probability of artificial and real nests, where I did not observe any predation events for passerine birds. Finally, a significant correlation was observed between lagged rodent abundance and artificial nest predation rates, although this relationship appeared to be largely driven by results from a single region (Ifjordfjellet). Nonetheless, the effect suggests that rodent abundance is a key factor to consider when assessing nest predation dynamics.

## 4.1 Evaluating how two types of camera traps affect predation probability on artificial nests: Brinno vs Browning

This thesis compares predation probability on artificial nests monitored using two different camera techniques. In Henden et al. (2024) 56 out of 60 nests monitored by a motion-sensor triggered Browning trail camera mounted on a pole, experienced predation, compared to the 19 of 39 nests without camera monitoring (control). In contrast, this study with ground-placed, camouflaged Brinno cameras with continuous recording every five seconds, only three out of 20 camera monitored nests were predated, while four predation events were observed on the 20 nests without camera monitoring (control). Results from the generalized linear model (GLM) indicated that artificial nests monitored by a close-up and ground-placed Brinno camera were less likely to be predated than those monitored by the more visually prominent Browning trail cameras on poles. Although the artificial nests monitored by Browning cameras had a longer exposure time (14 days compared to seven days for nests monitored by Brinno cameras), this effect has been considered in two different models (days monitored as offset model and the predictor model). The difference in predation probability between the two cameras became less pronounced with longer exposure periods, but that is also the case for the control group. The factors affecting the difference in predation probability, are likely camera size, camouflage and ground-placed vs prominent 0.5m poles. As the Browning Spec Ops Extreme used no-glow technology, it is unlikely that the infrared light attracted predators. Ground-placed and camouflaged Brinno cameras did not seem to inflate the predation risk of artificial nests. These findings suggest that this method might be a less biased way of monitoring artificial nests.

This study only identified two different predators, the long-tailed skua and the Arctic skua. Interestingly the high proportion of skuas as nest predators is inconsistent with previous findings, where ravens (*Corvus corax*) have been the dominant predator (Henden et al., 2024; Ims et al., 2013). A likely explanation is that the less visible Brinno camera reduced predation by visually dependent predators like corvids (Marzluff & Angell, 2007; Sonerud & Fjeld, 1987). There is possibly a difference in foraging technique between skuas and corvids, where skuas search more randomly than the more sight, memory and cognitive dependant corvid (Sonerud & Fjeld, 1987). Consequently, once a corvid learns to associate a camera mounted on a pole with a food source, it may begin to actively seek out similar structures and remember specific locations where it previously found food. However, with only two of the seven predation events in the study identified it is insufficient to draw any conclusions concerning difference in predator proportion across the different nest setups, other than the drastic reduction in corvid predation on Brinno monitored artificial nests compared to the artificial nests monitored by the larger Browning cameras on poles.

## 4.2 Does predation probability on artificial nests reflect predation on real nests?

Furthermore, the study attempted to compare predation probability on real versus artificial nests. The likelihood ratio test showed a significant difference in predation probability between the three distinct categories (artificial, passerine, and non-passerine) overall. However, subsequent post-hoc tests did not detect statistically significant differences in predation probability between the artificial nest (Brinno monitored and control combined) study and either category of real nests (passerine or non-passerine). Similarly, generalized linear models (GLMs) using both the upper and lower estimates of predation events on non-passerine nests failed to detect significant differences. While many studies (e.g., Bentzen et al., 2017; Burke et al., 2004; Major & Kendal, 1996) have reported higher predation rates on artificial nests compared to real nests, this particular study did not find statistical evidence of such an increase with the artificial nest design used. This may be notable, given that artificial nests typically lack the camouflage provided by an incubating bird, which is an important factor in reducing predation risk (Meyer et al., 2020; Smith et al., 2012).

Interestingly, predation was observed on non-passerines but not on the passerines. Consequently, the estimated marginal means indicated a trend where non-passerine and artificial nests had similar predation pressure, while passerine nests showed a lower predation probability (with no predation events observed). This could be because the passerine nests were often more hidden by vegetation and inside tufts, and generally harder to detect than the more exposed non-passerine nests. Although both passerine and non-passerine nests are considered ground-nests, there was a distinct difference in their visibility and vulnerability to predation. Thus, there seems to be a greater similarity between the non-passerine and artificial nests, both in predation probability, nest appearance and size. These differences highlight the importance of considering species-specific characteristics when using artificial nests for monitoring.

The observation of few predation events on passerine nests aligns with previous findings that passerines, such as the Lapland longspur, are less affected by predator-prey interactions in the Arctic, unlike various non-passerine species (Duchesne et al., 2021). However, despite this, the Lapland longspur is still in decline, and the cause remains unclear. A study in the Hardangervidda plateau have partly explained this decline by later spring thaw in the Lapland longspur's habitat (Fjeldheim et al., 2023). The observation of higher predation rates on artificial nests in the Arctic (Kubelka et al., 2018), especially in high productivity areas (Ims et al., 2019), are worrying for non-passerine species as they might be expected to experience similar trends in predation pressure. Elevated nest predation for the non-passerines may contribute to reduced reproductive success and, over time, raise the likelihood of population extinction (Bennett & Owens, 2002).

## 4.3 The relationship between rodent abundance and artificial nest predation

The generalized linear model (GLM) proved a significant effect between the year after a rodent peak and the predation rate on artificial nests, which aligns with former studies linking a lagged effect of rodent abundance with an increased nest predation rate (Gilg et al., 2003). However, further testing did show a difference in effect between the regions. The GLMs showed that the overall positive effect between the lagged rodent index and artificial nest predation was largely driven by the Ifjordfjellet region. In Vestre Jakobselv there was generally a lower predation rate, while there was an overall higher predation rate in Komagdalen. Both GLMs for these regions failed to show a significant relationship between lagged rodent index and artificial nest predation rate. This regional difference occurs despite the similarities in vegetation and latitude. One notable difference that could explain this difference between the regions is the presence of a large-scale red fox culling program in Vestre Jakobselv and Komagdalen, which is absent in Ifjordfjellet (Ims et al., 2017). Additionally, the study methods require extended time spent within the quadrants, potentially attracting corvids to the nests. Also, there is the possibility that territorial pairs of ravens in Vestre Jakobselv and Komagdalen have learned to associate human presence with nest locations, whereas such behavior may not have developed in Ifjordfjellet. Although only one of the regions showed a significant effect, it remains important to consider the rodent cycle when studying nest predation (Hansson & Henttonen, 1989; Ims et al., 2013). Alternative prey hypothesis suggests that generalist predators, which typically prefer rodents, increase in number during rodent peak years, and switch to alternative prey, such as ground-nesting birds, when rodent abundance declines (Angelstam et al., 1984; Hagen, 1952; Hansson & Henttonen, 1989). Therefore, further observational studies on both rodent abundance and nest predation are needed to better understand this interaction and monitor how it develops with ongoing climate change.

## 4.4 Limitations and further research

Considering that this study consists of various datasets with different methodology, there are a few distinct limitations that need to be mentioned. The first limitation to highlight is the exposure time of the artificial nests in the study. To better compare the results of the fieldwork in 2024 with the findings of Henden et al. (2024), an exposure period of 14 days would be favorable. The problem with a 14-day monitoring time is the battery life of the Brinno cameras. One possibility would be to replace the batteries after seven days, though this would increase the likelihood of predators picking up on the human scent or visibility near the nests. A better strategy could be to add an external battery source to the camera, though this would make the camera installation more conspicuous for predators. Therefore, it is important that the external battery source is also well camouflaged. Additionally, the Browning study spanned two breeding seasons (2018 and 2019), whereas the Brinno study covered only one (2024), which is likely resulting in more robust findings for the Browning study. The rodent population was in a crash phase in 2024, in contrast to 2018 and 2019, which represented a population increase and peak year, respectively. Therefore, the Brinno experiment should be repeated across all phases of the rodent cycle to obtain more reliable results. The final limitation noted is the location of the artificial nests. The nest locations could have been the same as Henden et al. (2024) to allow for a more precise comparison. However, this could also give a higher nest predation risk as the corvids may remember the previous nest locations and associate them with a food source.

Furthermore, there is the unstable generalized linear model (GLM) comparing artificial, passerine and non-passerine nests. Firstly, there was a relatively small sample size for both the non-passerine (22) and the passerine nests (19). Also, the standard error for the passerines (when using the lower estimate of non-passerines) was extremely large (2470). This is due to a combination of small sample size and low variation in predation for the passerine category as none of the 19 nests experienced predation. Also, the study would have more certain findings if better techniques had been used for estimating hatching time (to rule out predation), for instance the floating test (Ackerman & Eagles-Smith, 2010). Although this issue can be resolved with a suitable camera monitoring technique for real nests (for instance the Brinno-method), such a method would clarify whether egg or chick disappearance is due to predation or successful hatching.

This thesis aspires to act as pilot for future camera monitoring on real nests, to be able to identify actual real nest predators without influencing the predation probability. A camouflaged and close-up Brinno-camera seems to be a step in the right direction. Future studies should aim to include a camouflaged external battery source to ensure the camera lasts the entire incubation period of ground-nesting birds. Real nests will also give a more accurate predation pressure, as there are differences between artificial and at least some categories of real nests. Due to the positive bias in predation rates observed by former studies (Henden et al., 2024), it was decided to use the Brinno monitoring method on artificial nests before being applied to real nests, to avoid further predation pressure on the vulnerable ground-nesting birds in the tundra. However, one should still exercise caution when monitoring real nests, as the process may very well negatively affect the hatching results of the birds. Due to corvids learning skills there is still a risk that they will learn that even the more hidden Brinno cameras is a source of food in the future (Sonerud & Fjeld, 1987).

# 5. Conclusion

This study highlights the challenges of nest predation monitoring and the limitations of current methods. Artificial nests, monitored by close-up and camouflaged Brinno-cameras, offer a less biased monitoring method than the Browning trail camera on poles. There are still questions about whether predation rates on artificial nests serve as a reliable indicators of true predation intensity on real nests. Predation rates on artificial nests were more similar to those of non-passerines, likely due to similarities in size, exposure, and appearance. Despite both being ground-nesters, passerines and non-passerines differed in visibility and exposure, demonstrating the importance of considering both nest size and resemblance to the species of interest when using artificial nests for monitoring. Although the artificial nests resembled non-passerine nests and showed similar predation rates, several key differences must be considered, such as the absence of an incubating bird and the bird's active selection of nest site. Through the testing of how rodent abundance affects nest predation, it was found that the risk of predation for ground-nesting birds is highest in the year following a rodent peak, due to predators' numerical response to the rodent index. Furthermore, this effect varied significantly across different locations.

These findings represent an advancement in the monitoring of nest predators, nest predation rates and thereby bird populations. Further studies should focus on a non-biased and minimally invasive camera monitoring (both in terms of visible footprint and activity close to the nests) on real nests. Camouflaged Brinno-cameras are, as of today, a good choice due to their apparent lack of impact on predation rates and ability to record predators. This camera method will, however, require a camouflaged, external battery source to last the entire incubation period of ground-nesting bird species. It is important to note that this method has not yet been tested in alpine tundra with very short vegetation, which is characteristic of rock ptarmigan habitat. Further testing using artificial nests in such environments would be valuable to more thoroughly assess the impact of Brinno cameras on nest predation rates. Future monitoring studies should be long-term both to document differences in predation rate in all parts of the small rodent cycle and to monitor trends in nest predation with ongoing climate change.

Increased nest predation on vulnerable ground-nesting birds in the tundra is one of the many severe consequences of climate change. This poses a significant threat to the global biodiversity crisis. Hopefully, this thesis can offer valuable guidance for future observational studies. Identifying the nest predators and understanding the actual magnitude of the predation rate will provide stakeholders with accurate knowledge enabling them to take appropriate measures in mitigating biodiversity loss in the Arctic.

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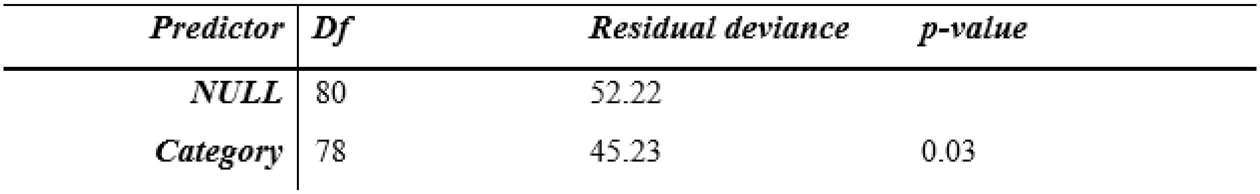
# Appendices

*Appendix A: analysis of deviance for an overall comparison of nest category*

Results from the analysis of deviance using a likelihood ratio test. The predictor 'nest category' significantly reduced residual deviance, indicating that nest type, overall, influences nest predation risk.



| ***Predictor*** | ***Df*** | ***Residual deviance*** | ***p-value*** |
|---|---|---|---|
| ***NULL*** | 80 | 52.22 | |
| ***Category*** | 78 | 45.23 | 0.03 |

*Appendix B: Rope dragging*

A method that allows fieldworkers to cover a larger area when locating ground nests. As the rope approaches the nest, the incubating bird flees, revealing the nest's position. Photo: Ekeli, J., 2024.

*Appendix C: Predation by an Arctic skua*

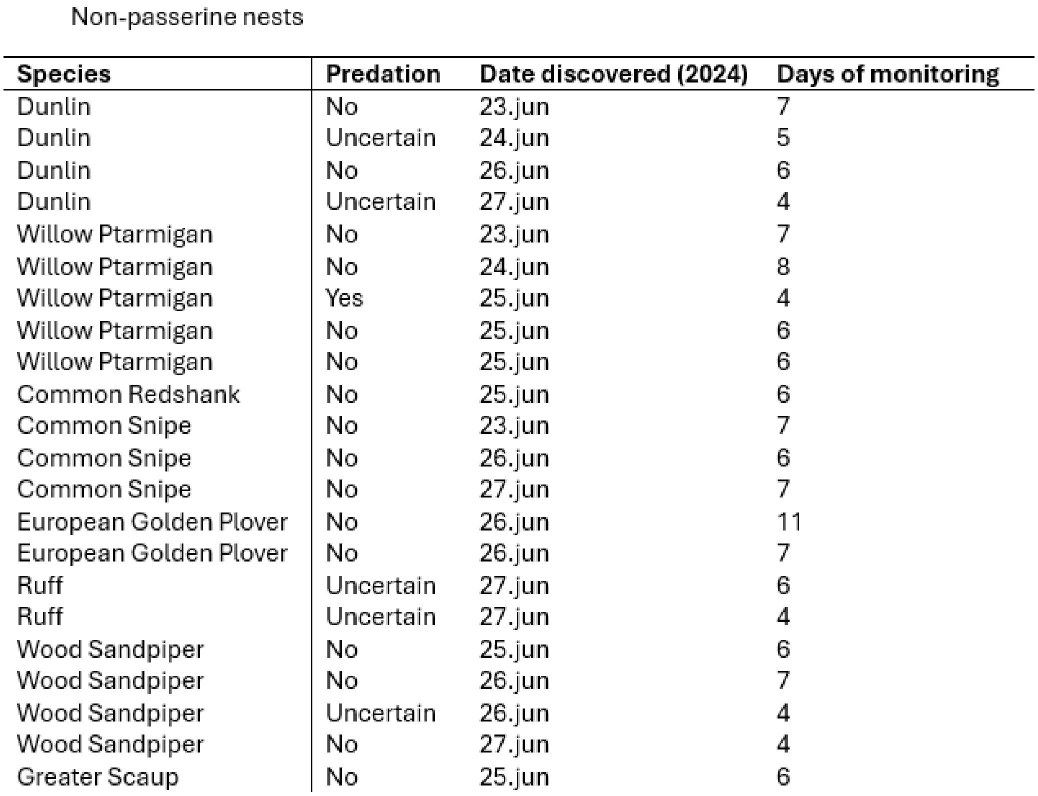
Predator identification of an Arctic skua at one of the artificial nests monitored by a Brinno camera. Still image from the surveillance footage.

*Appendix D: Long-tailed skua moving an egg*

A long-tailed skua moved an egg from one of the monitored artificial nests, without consuming it. The top-left image shows the nest before the event, while the top-right image shows it afterward. The bottom image captures the moment when the skua moves the egg, indicating that the predator could have consumed it but chose not to. In this study, the event was not counted as predation since no eggs disappeared. However, even if it had been considered predation, it would not have altered the results. Bottom photo: still image from the surveillance footage. The two top photos: Ekeli, J., 2024.

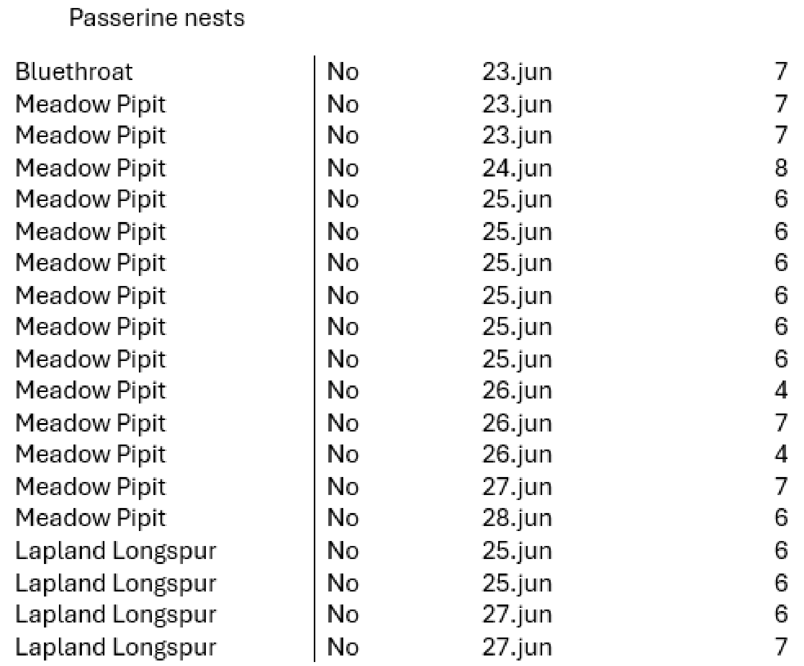
*Appendix E: The ground-nests located and monitored*

The non-passerine and passerine nests located and monitored in 2024. Included is the name of species, predation (no, yes and uncertain), date of discovery and days monitored. No passerine nests were predated.



Non-passerine nests

| Species | Predation | Date discovered (2024) | Days of monitoring |
|---|---|---|---|
| Dunlin | No | 23.jun | 7 |
| Dunlin | Uncertain | 24.jun | 5 |
| Dunlin | No | 26.jun | 6 |
| Dunlin | Uncertain | 27.jun | 4 |
| Willow Ptarmigan | No | 23.jun | 7 |
| Willow Ptarmigan | No | 24.jun | 8 |
| Willow Ptarmigan | Yes | 25.jun | 4 |
| Willow Ptarmigan | No | 25.jun | 6 |
| Willow Ptarmigan | No | 25.jun | 6 |
| Common Redshank | No | 25.jun | 6 |
| Common Snipe | No | 23.jun | 7 |
| Common Snipe | No | 26.jun | 6 |
| Common Snipe | No | 27.jun | 7 |
| European Golden Plover | No | 26.jun | 11 |
| European Golden Plover | No | 26.jun | 7 |
| Ruff | Uncertain | 27.jun | 6 |
| Ruff | Uncertain | 27.jun | 4 |
| Wood Sandpiper | No | 25.jun | 6 |
| Wood Sandpiper | No | 26.jun | 7 |
| Wood Sandpiper | Uncertain | 26.jun | 4 |
| Wood Sandpiper | No | 27.jun | 4 |
| Greater Scaup | No | 25.jun | 6 |



Passerine nests

| | | | |
|---|---|---|---|
| Bluethroat | No | 23.jun | 7 |
| Meadow Pipit | No | 23.jun | 7 |
| Meadow Pipit | No | 23.jun | 7 |
| Meadow Pipit | No | 24.jun | 8 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 25.jun | 6 |
| Meadow Pipit | No | 26.jun | 4 |
| Meadow Pipit | No | 26.jun | 7 |
| Meadow Pipit | No | 26.jun | 4 |
| Meadow Pipit | No | 27.jun | 7 |
| Meadow Pipit | No | 28.jun | 6 |
| Lapland Longspur | No | 25.jun | 6 |
| Lapland Longspur | No | 25.jun | 6 |
| Lapland Longspur | No | 27.jun | 6 |
| Lapland Longspur | No | 27.jun | 7 |

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