

Nest predation in birds of conservation concern

Case studies of monitoring and management

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Front cover:

Common eider (J. Stien) and Slavonian grebe (K.B. Strann) attend their nests.

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List of original papers

Paper I

Stien, J., Yoccoz, N.G. and Ims, R. A. 2010. Nest predation in declining populations of common eider (*Somateria mollissima*): an experimental evaluation of the role of hooded crows (*Corvus cornix*). *Journal of Wildlife Biology* (16), 123 – 134. doi: <http://dx.doi.org/10.2981/09-060>

Paper II

Stien, J. and Ims, R.A. Negative effects of human disturbance on nest success in declining eider colonies in northern Norway. In Review: *IBIS*

Paper III

Stien, J. and Ims, R.A. 2015. Management decisions and knowledge gaps: learning by doing in a case of a declining population of Slavonian grebe (*Podiceps auritus*). *Journal of Wildlife Biology* (21), 1, 44 - 50. doi: <http://dx.doi.org/10.2981/wlb.00026>

Paper IV

Stien, J., Strann, K.B., Jepsen, J., Frivoll, V. and Ims, R. A. Breeding persistence of Slavonian grebe (*Podiceps auritus*) at long-term monitoring sites: Predictors of a steep decline at the northern European range limit. In Review: *Journal of Ornithology*

Paper V

Stien, J. and Hausner, V.H. (manuscript). Volunteer hunter motivation and engagement to control the invasive alien American mink (*Neovison vison*) in Norway.

Abstract

In this thesis, I explored management options aimed at increasing breeding success of vulnerable species of ground nesting birds in Norway. Appropriate conservation management varies between species and between systems due to their differing and often complex abiotic and biotic interactions. To gain experience in working in this context dependent and multifaceted field, I used case studies of nest management of 2 species, common eider (*Somateria mollissima*) and Slavonian grebe (*Podiceps auritus*) in 2 different, but connected ecosystems; coastal and lacustrine.

In papers I to III, I explored the management actions of increasing nesting success in coastal nesting common eider and lacustrine nesting Slavonian grebe, and discussed whether these management actions could achieve the goal of halting the declines in these populations. In papers I and II, I investigated the reasons for nest failure of common eider. In paper I, I used the *a priori* hypothesis based on casual observation, that the hooded crow (*Corvus cornix*) was responsible for nesting failure, and carried out a crow removal experiment to test this. Using the results from paper I to define the scope of an observational study for paper II, I investigated the processes of nesting failure in common eider in order to find potential management actions that could increase nesting success in the study area. The combined results from papers I and II indicated that crows were the main proximate cause of nesting failure while human disturbance was the ultimate cause of nesting failure. Furthermore, the results indicated that decreasing nesting failure was potentially manageable by decreasing disturbance during nesting time and/ or guarding of nests while incubating females were absent from nests. These management actions should be tested in order to see if they can halt the declines in the local eider colonies.

In paper III, I used the *a priori* hypothesis that the invasive alien species American mink (*Neovison neovison*) was an important cause of nest failure of Slavonian grebe and carried out a series of experiments using an adaptive management approach. The hypothesis was based on a non-empirically assessed risk of factors pertinent to the decline of the Norwegian population and the observation that mink were present in the study system. The results from the study indicated that mink was unlikely to be a significant cause of nesting failure in the study system. The study further indicated a relevant framework for finding management solutions in the maze of management possibilities, where key system information and time to implement management is lacking. Based on the results from paper III, the knowledge that very few breeding grebe remained in northern Norway and availability of annual breeding data from a monitoring program in this region, I carried out a longitudinal study for paper IV in order to investigate pertinent factors in the apparent regional decline of the species. The results indicated that the grebe breeding population had declined steeply over the two last decades. Breeding

site persistence was positively related to the number of breeding pairs in the initial year of monitoring and in the final year of the data was negatively related to altitude. Testing of appropriate management is some way off, since there is a lack of understanding of the pertinent factors involved in population dynamics of the species in Norway or any other areas of its distribution. Furthermore, in Norway there is no monitoring of grebe population trends at the national level.

In paper V, I investigated the potential to engage hunters in large-scale and collaborative efforts to control mink populations in Norway. I used a questionnaire to examine the conservation support for-, and the level of efforts to remove mink expressed by mink hunters. I supplemented this with information gathered about current mink control projects and municipal paid bounty. The results suggested that state-led efforts are best concentrated around “conservation hotspots”, involving contract operators, bounty payments and awareness and recruitment programs. Few hunters were interested in mink control but those participating believe that mink is a conservation concern and particularly to ground nesting birds, suggesting that these popular species could be used to attract more hunters. As mink catch varies from low to high, control will probably require organization of hunter networks acting at an appropriate scale. Control projects could benefit from carefully targeted bounty payments as there was both a positive relationship between payment of bounty with the number of mink caught and was the main recommendation of hunters to make mink hunting more appealing to hunters.

1. Introduction

1.1 What are birds of conservation concern?

According to the IUCN Red List of Threatened Species, birds of conservation concern include species with a relative high risk of extinction. Criteria to assess this risk include small and / or declining populations with the decline assessed in terms of threshold proportional losses of their national or global population (Kålås et al. 2010). Depending on the scale of assessment, global or national, and the distribution of the species, holarctic or regional, the same species may have dual status as of conservation concern or not of conservation concern. In addition to the threat of imminent extinction, changes in ecosystem functioning and ecosystem processes due to large-scale declines of common species not yet applicable for Red List inclusion indicate that common species can also be of conservation concern (Inger et al. 2015). Thus, birds of conservation concern can be considered as potentially any bird population with a declining trend, with the decline being determined by the scale of the species population trend assessment.

1.2 Can nest predators cause declines in bird populations?

In stable populations, the number of sub-adults exceeds the breeding population, allowing a high mortality rate from multiple sources without leading to a decline in breeding populations. However, nest predation can be a significant determinant of population dynamics, especially in ground nesters (Martin 1993, Ibáñez-Álamo et al. 2012), and as such has the potential to both limit and regulate populations (Sinclair 1989, Newton 1998). Long-lived birds tend to have low annual productivity with occasional bonanza years, suggesting that low nesting success is of little consequence to population stability in these species (Gaillard et al. 2000, Coulson 2010). However, prolonged low productivity in long-lived species has been shown to result in declining populations (Hario and Rintala 2006, Reiertsen et al. 2013) suggesting that mortality due to predation may be important.

Generalist predators may have the ability to maintain prey populations at low numbers via prey switching (Redpath and Thirgood 1999, Šálek et al. 2004, Valkama et al. 2005, Begon et al. 2006, Matthiopoulos et al. 2007), thus preventing prey from escaping density dependent predation. This not only has an impact on the adult population but also on the reproductive effort of predated individuals. Furthermore, predation pressure by generalist predators may be exacerbated by anthropogenic effects (Schneider 2001, Neatherlin and Marzluff 2004). An extreme case is that of introduced generalist predators, as non-dependence on one particular prey combined with prey naivety, rapid predator population growth, high dispersal ability and density-dependent survival result in negative effects for many prey populations (Park 2004, Bonesi and Palazon 2007).

1.3 How should we investigate and mitigate nest predation in declining bird populations?

1.3.1 General approaches: Adaptive management and monitoring

Conservation involves the intervention in focal species ecology in order to prevent their decline and potentially to reverse this decline; i.e. management. Before we manage, we must decide what it is we need to manage. Should we modify the habitat in which the species resides, or should we modify some aspect of its ecological interactions (competition, predation), or should we do a combination of both in order to increase in population size? More often than not, we are faced with little or no knowledge of the pertinent processes in focal species ecology and at what life state the population is most sensitive to decline and/ or increase. We can learn about these processes from observational and experimental studies based on ecological theory and *a priori* hypotheses. We can use modelling, before empirical studies to reduce uncertainty in likely relevant parameters or estimates and underway incorporate new knowledge into these models. Observational studies may be informative in suggesting pertinent factors helpful to conservation management but lack the power of experiments to test whether manipulation is effective i.e. cause and effect. Effects measured in experiments can be informative to suggest management actions, but these effects are often not transferable to larger scale where management is practiced due to context dependent and cross scale effects (e.g. Castilla 2000). Furthermore, modifying the environment modifies the interactions with potential unforeseen consequences (Ruscoe et al. 2011, Norbury et al. 2013). Walters and Holling (1990) suggested the use of an adaptive management framework to inform management. This framework treats management actions as scientific experiments at management scale, allowing robust testing of effectiveness of experimental interventions and reducing structural uncertainty. Furthermore, it incorporates knowledge gained during the trials to modify the direction of future investigation and is considered a 'learning by doing' process as opposed to 'ad hoc' trial and error. Thus, it is a learning framework that can cope with dynamic (changing) systems. Learning is a two-phase (loop) process initialized by institutional learning also involving process learning, and an iterative phase involving monitoring, assessment and decision-making (Williams 2011).

Equally important as the spatial scale of investigation is the temporal scale, as salient factors change as uncertainty in the system is reduced. The adaptive monitoring framework (Lindenmayer and Likens 2009) incorporates the approach of the adaptive management framework and also emphasizes the importance of long-term monitoring designed to answer pertinent questions relevant to management. Research questions should be explicitly linked to management needs with the ability to incorporate learning into future direction and changing management goals. The introduction of the framework was a response to the wide-scale failure of monitoring programs to remain relevant to management objectives. Potential pertinent factors are explicitly represented by a conceptual model, thus

maintaining focus on the management objective. Adaptive monitoring shares the essential characteristics of adaptive management; i.e. reducing structural uncertainty by large-scale experimentation based on a priori based hypothesis and double loop learning. However, the double loop learning is in the evolution of questions designed and does not involve an institutional learning phase (Lindenmayer and Likens 2009).

Regardless of the approach used to investigate population decline, being able to monitor the response in question is vital in order to provide baseline data for measuring future change (observational study), or to measure the response to a perturbation or a human intervention (observational, experimental study, adaptive management and monitoring) (Lindenmayer and Likens 2009). In order to test management interventions, monitoring should be carried out both before and after the perturbation in an experiment design, the so called Before and After Comparison Investigation (BACI) design (Underwood 1994). In nest predator removal, a variety of monitoring methods exist for monitoring both the effect of removal on predators on predator numbers and the effect of predator removal on target populations, including observational counts, camera trap, or track traps (e.g. Graham and Lambin 2002a, Reynolds et al. 2004, Richardson et al. 2009).

1.3.2 Nest predator removals: An option for adaptive management of declining bird populations?

Predator removal has been successful in cases where nest loss due to predation is additive to other causes of mortality, and has led to increases in the number of hatching, fledging and breeding individuals (e.g. Chesness et al. 1968, Côté and Sutherland 1997, Bolton et al. 2007, Smith et al. 2010, Hanssen et al. 2013). These studies have been both small - and large-scale, however none have been published as, or mentioned as an adaptive management or monitoring approach. According to predator-prey theory (Holling 1959), the resulting increase in population size may lead to its escape from predator regulation, even when predators are no longer managed. Many predator removal studies fail to show such a population increase and attainment of new equilibrium. The lack of expected prey population response to predator removal can be due to compensatory predator mortality through inefficient management (Côté and Sutherland 1997) or changing systems leading to novel predator species, or other forms of compensatory mortality including disease and starvation (Camphuysen et al. 2002, Hario and Rintala 2006). Alternatively, lack of philopatry by breeding females could result in the lack of direct numerical increase (Pieron et al. 2013).

1.3.3 Finding solutions to implementing efficient management

Research-based management may well find salient factors underlying population decline and ways in which manipulation of these can be implemented to have desired objectives to halt the decline of target populations. However, by the very nature of the large scale of species distributions and the

number of species of conservation concern, implementation can be a considerable challenge. Control of established invasive alien species is a particular example where large-scale management is required and has become a responsibility for individual states (EU 2014). Invasive species are responsible for the loss of species diversity that in turn can lead to alteration and degradation of ecosystems (Baillie et al. 2004, Poorter et al. 2007, Simberloff et al. 2013) and are estimated to cost Europe at least 12.5 billion euros per year (Kettungen et al. 2008). Invasive alien species, such as mammalian predators, can exert negative effects via novel predation strategies on native prey species that have not had time to evolve successful predator avoidance strategies, which in many cases eventually have caused local extinctions (e.g. Atkinson 2001, Didham et al. 2005). Management of established invasive alien species is based on the goal of control, due to the recognition that distribution is too large and recolonization likelihood is too high to be able to eradicate the species at a national scale (EU 2014). This does not remove the requirement of management occurring at large scale, as small-scale action is likely to result in harvesting rather than control. Thus, inclusion of non-professional volunteers in a coordinated campaign can be a key method of attaining sufficient temporal and spatial scale and could significantly improve current invasive alien species removal (Gosling and Baker 1989, Bremner and Park 2007, Bell and Vanner 2011, Bryce et al. 2011).

One approach for control is for governments to invest in harvest incentives (i.e. *government based programs*). These may include i) *bounty programs* that give participants monetary rewards based on sufficient evidence of removal; ii) *contract operators* in which public or private organization are directly paid to remove or harvest the invasive species; iii) *recreational harvest* facilitated by training, education and outreach programs or by iv) encouraging harvest of the targeted species by *regulatory modifications* such as changing hunting seasons, licensing practices or bag limits.

As control of established invasive alien species is a long-term strategy, the retention of a sufficient volunteer pool is a key element for successful *volunteer based control programs*. Volunteers have diverse motives for participating in volunteer projects that also include personal benefits such as sociality and knowledge gain, in addition to the apparently obvious aiding conservation (Asah and Blahna 2012). In addition, deeper held beliefs and values may influence whether invasive alien species management is successful (Sharp et al. 2011, Fischer et al. 2014). Fulton et al. (1996) indicate that in western countries values regarding wildlife range along a continuum from domination to mutualism and that these values are important for explaining differences in acceptability of lethal control in particular. Thus, invasive alien species management may well recruit volunteers with different motives than other conservation stewardship programs. Investigation of volunteer motivations and values may therefore lead to tailoring of programs that are likely to be successful.

2. Scope of the thesis

In this thesis, I used 2 ground nesting species, the common eider (*Somateria mollissima*), a coastal breeder and Slavonian grebe (*Podiceps auritus*), a lacustrine breeder as case studies of the role of nest predation in population decline. Both species have large Holarctic distributions. However, the grebe was red-listed in Norway between 2006 and 2010 and there is still uncertainty over its population trend. The eider has experienced national declines and specifically the study colonies have experienced dramatic recent declines. A key aim of my study was also to investigate whether local predator removals could be effective for both species. In addition, I investigated the use of volunteer hunters in reducing the mink (*Neovison neovison*) predation (nest and adult) of ground nesting sea birds on a national scale. The specific questions I addressed were:

1. Does removal of hooded crow (*Corvus cornix*) result in an increase in common eider nest success at declining eider colonies?
2. What are the processes determining nest predation at declining common eider colonies?
3. Does a reduction in American mink increase the nesting success of Slavonian grebe?
4. What factors are involved in the decline in persistence of Slavonian grebe breeding sites at the northern edge of its range?
5. Do volunteer hunters have the potential to aid state managed mink control in populations of ground nesting sea birds, and how?

3. Study design and sampling methods

3.1. Papers I and II

3.1.1. Study area for papers I and II

The study area for papers I and II consists of two small low-lying islands Grindøya and Håkøya situated 2 km from each other along the coast of northern Norway at 69° 38 'N, 18° 52 'E and 69° 39 'N, 18° 49 ' E (Figure 1). The whole of the Grindøya (65 ha) was used while on Håkøya 64 ha of a total 361 ha was used. Both islands are located 2 - 3 km from urban areas of the city of Tromsø. Grindøya is a nature reserve with the largest concentration of breeding eider in the vicinity of the city. Both islands are low-lying with mosaics of open and wooded areas consisting of heath, mire and mountain birch (*Betula pubescens*). Håkøya has in addition, areas of grazed farmland and a settlement of approximately 60 dwellings (in 2006), whilst Grindøya has 3 coastal cabins that are now seldom used. Sheep have grazed the entire island of Grindøya annually between early June and the end of October since 2007. Access to the general public is limited on Grindøya between 1 May and 30 June to limit disturbance to ground nesting birds, particularly eider.

Both eider colonies are in long-term decline. The population on Grindøya has decreased steeply from over 500 pairs in 1995 to c.150 pairs in 2009 and onwards (Erikstad et al. 2010). Adult female survival decreased from > 80 % between 1986 – 2002 to a low of 53% in 2005 with the decrease in 2002 and continued low survival being attributed to mink predation of incubating females (Erikstad et al. 2009, 2010). Nest failure is as high as 62 % (Stien et al. 2010). The Grindøya colony has been the site of annual long-term monitoring since 1985 by NINA (Norwegian Institute for Nature Research) and is a key monitoring site in the Norwegian seabird monitoring program (SEAPOP). Data regarding Norwegian mainland vital rates for eider come from this colony, and the population is the source for a host of ecological, evolutionary and behavioural studies (Erikstad et al. 1998, Hanssen et al. 2002, Hanssen et al. 2003d, Hanssen et al. 2006). It is also a key site in terms of education, providing data for several PhD and Masters studies. The Håkøya colony has been the focus of low intensity annual nest monitoring since 2006. The colony is a remnant of an active eider down colony with an estimated population of over 600 nesting individuals (Olsen pers. comm.) in the middle of last century. The study area contained at least 49 nesting individuals in 2006, which decreased to 26 in 2011. Individual nest fates are followed but not individual females. Disturbance from other human activity was limited during the study period 2006 – 2011 as the colony area was little used during the study period. Nest failure is up to 39 % (Stien et al. 2010).

Predator communities on Grindøya and Håkøya are fairly similar, consisting of mostly large gulls (herring gull *Larus argentatus* and to a lesser extent greater black-backed gull *L. marinus*), hooded

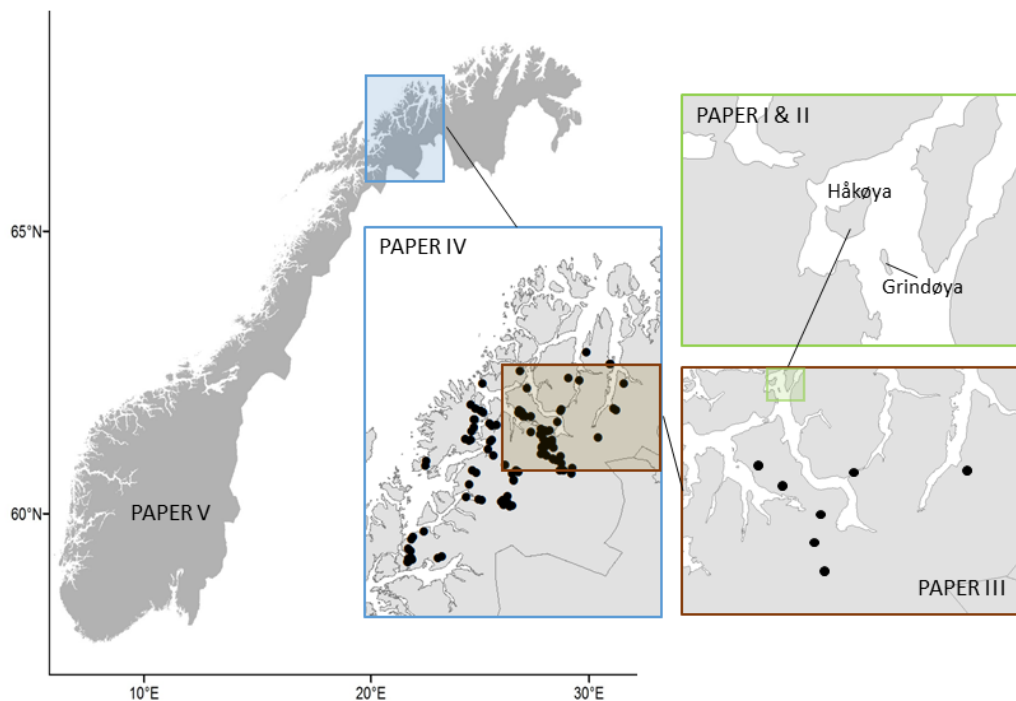


Figure 1. Study areas for the five papers in the dissertation. PAPER I & II show the location of eider colonies for the investigation of nest predation; PAPER III shows the location of Slavonian grebe breeding lakes for the investigation of mink predation; PAPER IV shows the locations for investigation of the decline in persistence of Slavonian grebe breeding lakes; and PAPER V shows the extent of mainland Norway, the scale used for the investigation of volunteer collaboration predator control management.

crow, raven (*Corvus corone*), white-tailed eagle (*Haliaeetus albicilla*), otter (*Lutra lutra*), mink (*Neovison vison*) and in addition on Håkøya, stoat (*Mustela erminea*). The occurrence of mink is sporadic, having been recorded in the Håkøya colony in 2006 and in the Grindøya colony from 2002 until 2009. The avian predators are nest predators apart from white-tailed eagle that takes adult birds on the open sea. Of the mammalian predators, stoat is a nest predator, mink predated both incubating females on the nest as well as eggs and young, while otter has not been observed predated adults or nests at either colony. At the time of the study initiation in 2006, hooded crow and herring gull were assessed as the most important nest predators (Erikstad and Tveraa 1995)

3.1.2. Study design for papers I and II

The whole of Grindøya was used in our study as it was combined with nests used for research studies on other aspects of eider breeding biology by the Norwegian Institute for Nature Research (NINA). On Håkøya, we used the northern part of the island as it holds the majority of the eider colony (J. Stien, pers. obs.) and used a similar area to Grindøya (64 ha and 65 ha, respectively). Protocols on Grindøya

followed the basic monitoring protocols used for long term monitoring by NINA researchers and nest search effort was coordinated in order to reduce disturbance to nesting birds. Research activity was intensive involving a high degree of disturbance especially during egg laying and hatching. Protocols carried out on Håkøya involved minimal disturbance as the goal was to observe nesting success only. Nests were searched for between 15 – 22 May (the onset of the breeding season varied slightly between years) and 5 June. Monitoring of nesting outcomes carried on until 28 - 30 June. The initial clutch size was used as a proxy of investment and or/ likelihood of survival at time of nest discovery. Maximum clutch size was recorded on Grindøya by revisiting the nest every 2 - 3 days until no new eggs were laid (Erikstad et al. 1993) and on Håkøya by revisiting the nest after calculating the revisit date as that allowing for a maximum of 6 eggs to be laid (Watson et al. 1993).

Study design specific to paper I

We used a BACI (Before and After Comparison Impact; Underwood 1994) design to investigate the effect of crow removal on nesting success. The nesting success of eider nests was monitored at both colonies in each of the study period years 2006 to 2008. In 2007, crows were removed from Håkøya and in 2008 were removed from Grindøya. The effect of crow removal on crow activity in each year was measured by carrying out paired colony watches of crow activity.

After eider egg laying was complete, the nests on Håkøya were monitored every second day in order to try to identify predator species. Nests on Grindøya were monitored up to 6 times after maximum clutch size was observed. These visits included catching and ringing on days 5 and 20 of incubation and checking for hatching success after day 20 of incubation. Nests of unknown lay date on Grindøya received *ad hoc* visits in 2006 and 6 visits between June 3 and 30 in 2007 and 2008. We followed 543 eider nests to completion (i.e. where at least one egg hatched) and used the data to estimate nesting success. We used a logistic exposure model to estimate daily nesting success, with the interaction between colony and year as the predictor and colony, year, Julian day, initial clutch size (t_0) and maximum clutch size as covariates.

The number of territorial crows was recorded at both colonies in all years of the study, using transects spaced at 80 m intervals to locate territories and thereafter the territories were searched to locate the nests. Transects were walked between 24 April and 16 May (the onset varied slightly between years). On Grindøya, transects started on 12 May due to logistical constraints of reaching the island earlier. Larsen Traps were used to trap territorial pairs of crows. We assessed the effectiveness of crow removal by monitoring crow activity at both colonies using paired counts (carried out in successive days at each colony and at the same hour) over a range of times during the breeding season. We used

log-linear regression with Poisson distribution and interaction colony*year to predict the number of crows at each colony.

The activity of other predator species was monitored in all years by recording aboreal nesting predators during eider nest searches, and gull numbers by two counts one at the end of May and one during mid – June. Stoat activity was recorded in 2007 and 2008 by weekly checking of tracking tunnels laid out on a grid system across the study areas between 12 – 16 May (Graham and Lambin 2002). Sixteen tunnels were used on Grindøya and 17 on Håkøya with positioning reflecting high-density areas of eiders.

Study design specific to paper II

We monitored the nesting success of a total of 1003 eider nests at both colonies between 2006 and 2011. At 205 of these nests, we deployed RECONYX™ cameras to allow detailed knowledge of predation events to be investigated at the nest level. At camera monitored nests, variables pertinent to eider nesting success were collected (Götmark 1989, Mehlum 1991, Hanssen et al. 2003a, Noel et al. 2005, Andersson and Waldeck 2006). These variables were initial clutch size, Julian day, nesting habitat, nesting cover, nearest neighbour distance and distance to the sea. Exploration of the camera data allowed complete nest leaving histories with nest leaving classified as either disturbance or ‘natural’ to be identified. Combining these nests with environmental variables gave a sample of 103 camera monitored nests to explore the effect of the contrast in disturbance rates between the 2 colonies. We used disturbance as a predictor of nesting success and natural nest leaving (recess) and the environmental variables as covariates. We controlled for the possible effects of camera deployment on nesting outcome (Richardson et al. 2009) by comparing nesting success of 41 paired camera and non-camera monitored nests at both colonies in 2009. We estimated the number of breeding crows and large gulls in all years using transects spaced at 80 m intervals over the entire area of each colony to estimate crow territories and counts during searches for eider nests to estimate gull numbers (pairs).

3.1.3. Study species for papers I and II

Common eider

The common eider is a well-studied sea duck that has been the focus of many evolutionary, physiological and ecological studies throughout its circumpolar range (e.g. Mehlum 1991, Erikstad et al. 1998, Desholm et al. 2002, Hanssen et al. 2003d, Hanssen et al. 2006, Hario and Rintala 2006, Öst et al. 2007, Coulson 2010, Wilson et al. 2012). Most studies focus on the females as males are not actively involved in nesting or raising of young (Steele et al. 2007). The eider has a circumpolar distribution with an estimated population of over 3 million individuals (Bird Life International 2015).

As such, it is listed as 'least concern' on the IUCN species Red List. However, the population has been in wide scale decline in recent years (Desholm et al. 2002, Hario and Rintala 2006, Coulson 2010, Wilson et al. 2012). Adult survival is typically high, breeding is delayed with first breeding from 2 years of age, and periodic non-breeding years occur due to high energetic costs of breeding (Yoccoz et al. 2002, Hanssen et al. 2003, Hario and Rintala 2006, Coulson 2010, Wilson et al. 2012). Females have high natal-, and general philopatry (Swennen 1991, Bustnes and Erikstad 1993, Hanssen and Erikstad 2012) and are short – distance migrants (Bustnes and Erikstad 1993, Camphuysen et al. 2002, Lehikoinen et al. 2006).

As a capital breeder, female condition is expected to play an important role in reproductive output and indeed recruitment. Female condition has been linked to large-scale climatic processes on the wintering areas (Lehikoinen et al. 2006, Descamps et al. 2010) and reproductive output is positively related to female condition with clutch size, hatching success and fledging success being positively related to female body mass (Erikstad et al. 1993, Erikstad and Tveraa 1995, Hanssen et al. 2003a). Further, recruitment has been linked positively to fledging success 3 years earlier (Hario and Rintala 2006). Impacts related to nest predation may also influence fledging success and recruitment, reducing the likelihood of breeding in up to 2 successive years after nest loss (Hanssen and Erikstad 2012). Nesting and fledging success may be highly variable between years (Noel et al. 2005, Hario and Rintala 2006, Coulson 2010, Wilson et al. 2012).

Eiders have a low clutch size (mean 4 eggs, range 1 - 6). They display both partial delayed onset of incubation and complete fasting during incubation, leaving the nest for short periods during incubation only in order to drink (Erikstad and Tveraa 1995). This behaviour is seen as a trade-off between food resource maintenance and predator defense (Andersson and Waldeck 2006). The predation rate of nests decreases from a high rate of 48 % for clutches of one egg to an average of 3 % from the second egg (Hanssen et al. 2002). Incubation lasts roughly 20 days (Erikstad et al. 1993). As a consequence of complete fasting, females may lose up to 40 % of their pre-nesting body mass (Parker and Holm 1990). Young are nidifugous and leave the nest after a few days, often to be cared for in crèches by several females (Öst et al. 2007).

Population declines in eiders have been attributed to reduction in adult survival (both on wintering and breeding grounds) due to unknown cause (Coulson 2010), large scale mortality due to starvation on wintering grounds (Camphuysen et al. 2002) and prolonged successive high annual mortality of ducklings due to outbreaks of viral infections (Hario and Rintala 2006, Hario and Rintala 2009). Predation of adults may also be important in causing declines (Erikstad et al. 2009). Population declines are associated with a sex-biased higher mortality rate of females, occurring both at the pre-breeding

and adult stages (Lehikoinen et al. 2008, Erikstad et al. 2010). Evidence for compensating mechanisms in eider populations are equivocal. Density dependence (acting on the fledging rate) has been demonstrated in long term studies in southern Finland, though was insufficient to halt a long term decline in the population (Hario and Rintala 2006). The decline was also not compensated for by a decreased age of first breeding, with food shortage and/ or non-lethal viral infections being suggested as reasons for late maturation (Hario and Rintala 2009). In long-term studies in north east England, density dependence has not been demonstrated at any life stage (Coulson 2010). In the light of population decline and biased ratios towards males, there is also no evidence of (facultative) manipulation of sex ratios at laying or hatching stages to favour increased production of female offspring (sensu Fisher 1930).

Hooded crow

The hooded crow is a common generalist predator with a wide distribution across north, eastern and southeastern Europe and the Middle East (Coombs 1978). No estimate is available for its worldwide population size as it was until recently considered as a subspecies of the carrion crow (*Corvus corone*), and as such has been assessed as having lowest concern on the IUCN Red List (BirdLife International 2014). In Norway, the population of hooded crow is estimated to be between 200,000 and 600,000 pairs and outside of the breeding season can be legally killed. Hooded crow has delayed maturity, not breeding before 2 years old (Coombs 1978), with annual survival rate estimates of adults ranging between 48 % and 70 % (Haukioja 1969, Holyoak 1971, Loman 1980). Productivity defined as the mean number of fledglings per clutch varies between 0.72 and 3.20 (Zduniak and Kuczyński 2003). Densities of nesting pairs vary, being highest where food sources are concentrated and range between 1.6 – 1.8 km² for island habitats (Loman 1980, Erikstad et al. 1982, Parker 1985) and 0.3 – 9.0 pairs km² in rural areas (Loman 1980). The species has been recognised as a benefitting from anthropomorphic subsidies (Marzluff and Neatherlin 2006). Nesting habitat is primarily boreal, with breeding occurring from early spring (depending on altitude) and lasts approximately 75 days from nest building to fledging of young (Coombs 1978, Zduniak and Kuczyński 2003). Crows are smart, demonstrating good cognitive ability in obtaining and securing food resources (Picozzi 1975, Sonerud and Fjeld 1987, Sonerud 2001, Neatherlin and Marzluff 2004, Marzluff and Neatherlin 2006, Melletti and Mirabile 2010). Removal studies, both experimental and observational indicate that crows may have a negative impact on nesting success but often do not due to compensatory predation by other predator species (e.g. see review in Madden et al. 2015). Thus, though appealing, this management action may be costly and fail in its goal.

3.2. Papers III and IV

3.2.1. Study area for papers III and IV

The study area for paper III is located between 69° 04' and 69° 24' N and 18° 39' and 20° 20' E III and is a sub area of the study area for paper IV which is located between 68°30' and 69°43' N and 16°39' and 22°09' E (Figure 1). Located in northern Norway in Troms and Nordland counties, the study areas consisted of 7 and 104 breeding lakes for Slavonian grebe in papers III and IV respectively. All sites were part of a long term monitoring of annual breeding success project, carried out annually since 1991 by researchers at NINA and were therefore active at the start of monitoring.

The area forms the northern range of the European population of Slavonian grebe (Fjeldså 1973a) (as well as quite possibly the northern range for the global population (Fournier and Hines 1999)) and was the former core of the breeding population in Norway. Water bodies between 0 m and 269 m above sea level are used as breeding sites. All sites were ice covered during the winter with ice melt varying between late May and mid-June. All sites were inland and fed by streams or rivers and/ or had rivers as outflows. Average (mean) water body area was 93 ha (median 19.18, range 0.34 – 1521 ha) and mean altitude was 90.98 m (median 91.00, range 0-269 m). Immediate (within 200 m of lake edges), surrounding vegetation was dominated by mosaics of mountain birch, Scots pine (*Pinus sylvestris*), mire, heath and grassland. Agricultural grassland also existed around some lowland lakes. Lake bedrock was mostly calcareous rock, with smaller frequencies of marble and occasional granite. Lakes were mostly oligotrophic with several mesotrophic and eutrophic lakes. Dominant shallow water vegetation included bottle sedge (*Carex rostrata*) and to a lesser extent bogbean (*Menyanthes trifoliata*), and provided nesting habitat for the grebe. Lake vegetation was sparse in oligotrophic lakes forming small pockets of nesting habitat, and more or less continuous in eutrophic lakes, providing continuous nesting habitat around the lake edge perimeter.

At the beginning of the monitoring period in 1991, 104 sites were occupied giving a total of 405 pairs (mean number of pairs per lake at the start of monitoring was 3.91 ± 0.24). By 2012, the site occupancy for the whole area had decreased to just 25 % of that in 1991 and the number of breeding pairs had decreased to 49 (mean 0.50 ± 0.10 per site, $n = 104$). A national action plan for conservation of the species assessed that decline was most likely due to a range of negative effects during the breeding period and singled out predation by mink during the nesting period as a probable factor in the decline of the Norwegian population (DN 2009). In addition to mink, other common species of the nest predator community were hooded crow, heron (*Ardea cinerea*), common gull (*Larus canus*) and otter.

3.2.2. Study design for papers III and IV

Study design for paper III

The study took place between 2010 and 2011 with 7 lakes being chosen in the same region and near each other so as to represent a single management area. Lakes were expected to be independent with respect to individual mink during their breeding season, with Euclidean distance (range 5 – 43 km) between study lakes being greater than reported mean home range for territorial males during the breeding season in freshwater habitats (Gerell 1970, Dunstone and Davies 1993). Three sequential experiments were undertaken.

Experiment 1

The first experiment involved a paired treatment and control BACI design at five lakes, whereby the activity of mink before and after trapping events was compared to the activity of mink at monitored control lakes in order to assess the effectiveness of mink passive trapping throughout the ice-free period in 2010. At each lake, 6 mink rafts were deployed at 1 km intervals to monitor mink activity (Bryce et al. 2011) weekly between ice melt (27 May – 9 June) and re-freezing (10 October). At the treatment lakes, traps were deployed initially after mink activity was recorded and from mid-July, permanently to increase chances of capturing mink. We expected mink to be active at all sites and that removal during the grebe breeding season would result in a decrease in mink activity.

Experiment 2

The second experiment proceeded in the same treatment and control lakes in early spring between 2 and 7 April 2011, while lakes were still frozen and before the arrival of the grebes. A transect was walked along each lake edge using dogs to track mink activity between 2 and 4 April. In treatment areas, discovered mink holes were excavated to remove mink and additional traps were laid and checked up to 5 days later to increase the chance of capture after activity had been recorded. We expected mink to be active at all sites and that the removal of mink would result in a decrease in the activity of mink.

Experiment 3

The third experiment focused on assessing the predation risk on grebe nests in the following breeding season in 2011. It involved camera monitoring of artificial nests in grebe nesting habitat at 7 lakes (the original 5 plus 2 additional lakes). Five pairs of nests were deployed at roughly 250 m intervals along a 1 km transect between 22 -23 June and 14 July. The first of each paired nest was equipped with an Acorn game monitoring camera (Little Acorn 5210A) while the second had none and was used to assess the effect of cameras on predation rates. Variation in predation rate between the 7 lakes was analysed with a simple logistic regression model in R with binomial distribution. The predictor variables area,

altitude, river length and distance to nearest stream were entered singularly. We expected that mink would be the main predator at the artificial nests.

Study design for paper IV

We analysed data from 104 breeding sites monitored annually between 1991 and 2012. Monitoring was based on two visits each year in the period 1991-2012. The first visit was around 22 June, roughly 3 weeks after ice melt and the second between 10 and 20 July (exceptionally the end of July). Number of nesting pairs, territorial pairs and non-territorial individuals were counted in both visits from standardised observation points using binoculars and telescope. The counts of nesting pairs were used in the analysis and were expressed as a single unit of observed number of breeding pairs per lake.

Variables pertinent to breeding success were used as predictors in Cox proportional hazards models to investigate persistence in the use of breeding sites. Persistence was defined as the number of years from the onset of the monitoring to the year grebes disappeared from the site or until the end of the monitoring series in case grebes continued to be recorded during the entire 22 year monitoring period (i.e. until 2012). In the absence of lake specific data, all predictive variables apart from number of breeding pairs were proxies for habitat quality or factors that could affect breeding numbers negatively.

3.2.3. Study species for paper III, IV and V

Slavonian grebe

The Slavonian Grebe, hereafter referred to as grebe, has a circumpolar distribution mainly at 50 – 65 °N in the boreal climatic zone, breeding in North America, Europe and with isolated populations in Iceland, Færoes and Scotland (Bird Life International 2011). The general trend for the population is declining but due to the size (140,000 - 1,100,000 individuals) and geographical extent of the population, the species is categorised as ‘least concern’ on the IUCNs red list (Bird Life International 2011).

There are no detailed estimates of adult survival or age of recruitment for the grebe, however unpublished data indicates that individuals return to breed in their second year (T. Lindberg pers. comm., S. Benn pers. comm.). Breeding success ranges from 0.20 – 1.06 fledglings (large young) per pair with a mean value of 0.58 (Ewing et al. 2013). The grebe spends most of the year in marine habitat but migrates inland to breed between May and September. Breeding can occur in both freshwater and brackish water and in a wide range of lakes sizes, with sites (< 10 ha) common in north America and the Baltic and a larger range of site area used in northern Norway and Iceland (Fjeldså 1973c, Faaborg 1976, Ulfvens 1988, Ewing et al. 2013). Sites commonly have between 1-2 pairs and seldom more than 20 pairs per lake (Fjeldså 1973e, Faaborg 1976). In Norway, winter habitat is in coastal archipelago and

outer fjord systems (Fjeldså 2004) with part of the population migrating as far south as the Scottish coast (Aarvak and Øien 2009). Onset of nest building is determined by ice melt and varies considerably with latitude, altitude and season (Cramp et al. 1977, Fjeldså 2004). Nests consist of floating rafts of dead plant material, constructed in shore vegetation. Diet during the breeding season consists mostly of fish by biomass but also of aerial and aquatic invertebrates (Fjeldså 1973c, Dillon et al. 2010). Young and adults migrate to the coast in September.

The Norwegian population forms the northern range limit for the species in Europe (Fjeldså 1973a). Historical records indicate that the core area in the 1970's was in northern Norway and was estimated to be c. 400 pairs (Fjeldså 1980). Although no systematic monitoring of grebe occurs on a national scale, regional scale monitoring of core sites in Troms and bordering Nordland reveals a decrease in the use of breeding sites compared to when monitoring began in 1991 (Strann and Frivoll 2010, Strann et al. 2014). National declines have been reported in neighbouring countries with an estimated 54% decline between 1972 and 1996 in Sweden (Douhan 1998) and strong negative population change index since 1997 in Finland (Pöysä et al. 2013). In Sweden, the population appears to have increased again and in 2011 was estimated to be close to the 1972 estimate of 2200 pairs (Norevik 2014). This increase has been accompanied by an apparent eastward shift in its range away from inland areas to areas along the Swedish Baltic coast (Norevik 2014). Declines in grebe populations have been associated with continued low breeding success (Ewing et al. 2013), deterioration in nesting habitat and increasing availability of new habitat (Douhan 1998). The Icelandic population is increasing and reproduction is generally higher than reported for European and American studies (T. Lindberg unpub.), without the drivers of the population dynamics being known.

American mink

American mink, hereafter referred to as mink is an invasive alien species in 28 countries in Western Europe, having established wild populations from escaped individuals from fur farms or, in Russia from purposeful releases for hunting (Bonesi and Palazon 2007). Mink predation causes negative effects on many native prey species in European countries and as such it has been a target of several control campaigns (Bonesi and Palazon 2007).

The species inhabits semi-aquatic coastal and inland wetland systems (Halliwell and Macdonald 1996, Ahola et al. 2006), living at higher densities in coastal habitats than inland habitats (Gerell 1970, Dunstone and Birks 1983, Helyar 2005), presumably due to the greater availability of prey in coastal habitats (Dunstone and Birks 1987, Bonesi et al. 2000). Mink are generalist predators eating a wide range of aquatic and terrestrial prey (Dunstone and Davies 1993a).

Males are polygamous, with territories overlapping those of several females (Yamaguchi et al. 2004). While males range widely during the breeding season, female territories are established by late winter (February – March). Females reproduce once a year, in spring (April - June), are central foraging feeders for 5 weeks while kits are young (Gerell 1971, Dunstone and Davies 1993a, Yamaguchi and Macdonald 2003). During autumn, territories break up and family parties and adult males roam over large areas (Dunstone and Davies 1993a, Yamaguchi and Macdonald 2003). Dispersal occurs mostly along wetland corridors (Gerell 1970, Birks and Linn 1982, Bryce et al. 2011) but can also occur over land. In coastal habitats in Fennoscandinavia, dispersal is aided by winter ice (Kauhala 1996). Mountain chains and stretches of open water over 2 km appear to be effective barriers or filters to encroachment on new areas (Kauhala 1996, Craik 1997, Zalewski et al. 2009). Effective dispersal results in rapid establishment of new populations (Bevanger and Henriksen 1995, Bonesi and Palazon 2007).

Mink populations in several countries appear to have peaked and are now on a decline (Bonesi and Palazon 2007), suggesting that the invasive population has reached carrying capacity. Rapid colonisation of recently cleared areas indicate that there is still a surplus of individuals produced in neighbouring areas and that density dependent processes may operate. The occurrence of 7 year population peaks in Iceland suggest delayed density dependence may also be a feature of mink population dynamics (Einarsson et al. 2006). The occurrence of density dependence makes control programs more difficult as removal results in increased growth rate of the remaining population (Pasko et al. 2014). Source areas are probably coastal habitat, with dispersal to inland areas, at least at a small scale (Bodey et al. 2010). There is little information on either survival or age of first breeding for mink.

As an invasive species, mink has few natural predators and competitors. Red fox (*Vulpes vulpes*), white-tailed eagle and eagle owl (*Bubo bubo*) predate mink in unknown numbers. White-tailed eagle may also cause mink to alter feeding patterns and area use, and otter can outcompete mink (Bonesi and Macdonald 2004, Salo et al. 2008). However, otter and mink can co-exist, especially in coastal areas (Christensen 1995, Harrington et al. 2009).

No systematic monitoring of mink or the effects of mink removal on mink numbers and focal prey species have been undertaken in Norway. However, casual observations clearly indicate that mink can have a devastating effect on ground nesting seabirds (Stien et al. 2011). There is also a considerable population in Norway as between 5100 and 6700 mink were reported caught between 2002 and 2013 (SSB 2013b). Several grass roots initiatives in coastal areas suggest that successful removal of mink leads to an increase in focal prey species (Stien unpub.). Key factors in the successful removal of mink appear to be removal after territory establishment (Craik 1995), removal on a scale that is larger than

the dispersal distance of a mink (Helyars 2005, Bryce et al. 2011) and removal over repeated years to remove reinvasions from neighbouring populations.

3.3. Paper V

3.3.1. Study design for paper V

We made a questionnaire (Questback) available via the internet to all members of the Norwegian Association of Hunting and Angling (NAHA) regarding the 2012 hunting season to investigate the prospects of engaging hunters in large scale and collaborative efforts to control mink populations in Norway. Specifically, we investigated the potential for establishing harvest incentive based mink control.

Hunting effort

The variables belief, motivation, and whether hunters received bounty payments were used to predict hunting effort which was measured by the 2 dependent variables number of mink caught-, and number of days hunting during 2012. We used hierarchical agglomerative clustering to create classes of belief, motivation and leadership and log-linear regression with Poisson error to estimate hunting effort. We predicted that consumptive motivated hunters would catch larger numbers of mink than those with more appreciative motives. We also expected that hunters receiving bounty payments would catch more mink than those not receiving bounty payments due to the incentive of economic reward. We included beliefs as a covariate in order to control for heterogeneity in respondent beliefs.

Conservation attitudes

We expected mink hunters with more utilitarian wildlife values and who hunted primarily for consumptive reasons to be more oriented towards the protection of fish and game species (sensu Kaltenborn et al. 2012). Mink hunters expressing stronger caring beliefs for animals were expected to shoot or trap mink primarily because of biodiversity protection (i.e. Red List species). We examined the relationship between beliefs, motivations and conservation attitudes using cross tables.

Leadership

We used hierarchical agglomerative clustering to form the classes of desired leadership. We used cumulative logistic models to predict the effects of belief and motivation on predict leadership classes. We expected hunters with social and/ or recreational motives to favour government leadership and included beliefs as a covariate in order to control for heterogeneity in respondent beliefs.

Recommendations

We included an open question enabling hunters to recommend ways to make hunting easier or more appealing. We used these answers to support the results from the above analysis.

4. Results and discussion

4.1. Results and discussion for paper I and II

The nesting success of common eider increased significantly during the crow removal on Håkøya in 2007, but remained constant and low during the crow removal year on Grindøya in 2008. Thus, the hypothesis that the removal of a single nest predator species would result in an increase in nesting success was only partly supported. Equivocal effects of nest predator removal as a management option to increase nest success have been shown in other nest predator removal studies (e.g. Côté and Sutherland 1997, Bolton et al. 2007). Lack of a constant positive effect has been shown to be due to incomplete removal of targeted predator species or compensatory mortality from increased predation of non-targeted nest predator species in multi predator systems (Crabtree and Wolfe 1988, Sargeant et al. 1995, Bolton et al. 2007). Alternatively, mortality due to crow predation could have been compensated by alternative predator species. On Grindøya, there was evidence of the effects of management failure, with new nesting pairs of crow moving into vacated territories and foraging by crow pairs from neighbouring islands. We could also not exclude the possibility of compensatory predation from large gulls on Grindøya as they are also common nest predators of eiders (Milne 1974, Götmark 1989). The results from Håkøya indicated that the effects of crow removal on nesting success can be seen after cessation of management in common with other removal studies (e.g. Parker 1985). The length of this effect is likely to be dependent on prey densities and immigration rates from neighbouring areas.

The results from study I also indicated consistent and contrasting nesting success at the two colonies during the study period, with consistently high nesting success on Håkøya and consistently low nesting success on Grindøya. This could be due to inter-colony differences in eider nesting density or habitat, affecting predation rates, or differences in predator species and/ or abundances at the two colonies. Alternatively, nest loss at Grindøya could be mediated by differences in human activity at the two colonies or inherent differences in eider body condition affecting rates of nest abandonment. In either case, predation would only be the proximate cause of nest loss. The results from study I highlight that proximate and ultimate causes of nest predation are context dependent and need to be investigated at the focal scale of interest, where focal scale refers both to scales of space and time. It also indicates the usefulness of before and after comparisons and experimental design in determining appropriate management.

Study II enabled investigation of the processes involved in nest predation in the 2 two eider colonies and the unravelling of sources of context dependency that could cause the contrasting nesting success. Furthermore, the results identified management actions that can increase nesting success at both

colonies. The analysis of a 6-year time period reinforced the finding of Study I that the contrasting nesting success was indeed a characteristic of the two colonies with nesting success being significantly higher at Håkøya than at Grindøya. The frequency of nest disturbance was much higher at Grindøya than Håkøya and this led to a steeply increased risk of nest failure due to predation. Thus, the results are consistent with the hypothesis that the lower nesting success at Grindøya is ultimately due to a constantly higher rate of human disturbance of nests at this colony. Negative effects of human activity on nesting success are well documented (e.g. Bolduc and Guillemette 2003, Ibáñez-Álamo et al. 2012) but not necessarily always occurring (e.g. Nisbet 2000). Thus, management of breeding populations exposed to human activity should include an assessment of effects of human activity on productivity. As the management aim of the two studies was to increase nesting success at the two colonies, the results from the two studies indicate that this can be achieved by reducing human disturbance and/ or after each disturbance event guarding of the nests until incubating females return. The subsequent step in management to achieve the goal of increasing nesting success would therefore be to trial these actions to see whether they are successful.

In addition to identifying plausible options for increasing nesting success in these two declining colonies of eider (i.e. decreasing disturbance and/ or implementing nest guarding) study II also indicated that observational studies with appropriate monitoring methods are relevant to indicate pertinent factors in processes. Conservation of declining species ideally involves not just halting the decline, but also increasing population size. The options from study II allow the testing of a hypothesis aimed at management action to increase colony size at these two specific colonies, that mortality at the nest stage is additive and therefore increasing nesting success will result in increased colony size via increased recruitment. This is a big supposition for a species with nidifugous young and a subsequent stage of delayed maturity that is exposed to many forms of mortality (Hario and Rintala 2006, Coulson 2010). However, a link between high nesting success and population increase is aided by the high natal philopatry of the species (Swennen 1991) and the high breeding philopatry which has been found at the Grindøya colony (Hanssen and Erikstad 2012). As the potential benefits of increasing the stability of the local distribution of the species are high, these management options should be tested. Thus, the stepwise learning from an experimental study (study I) and an observational study (study II) provide relevant management actions aimed at increasing the robustness of two local populations. Although the results are context dependent, the stepwise learning at local scale is an approach that is highly applicable to conservation management in general.

4.2. Results and discussion for paper III and IV

Studies III and IV demonstrate how a combination of systematic methods at different scales can be informative for management of focal conservation species. We found no support for the hypothesis that mink was a significant negative factor on breeding grebe in study III. An adaptive management framework allowed us to conclude this in a short time period from a start point of no knowledge about mink ecology in Norwegian lakes and no knowledge about the reasons for the regional decline in grebe. The study thus prevented the implementation of costly inefficient and/ or inadequate management aimed at removing negative factors on breeding success of grebe in the study area, even though the removal of mink *per se* is an appealing idea due to its invasive alien predator species status. The adaptive management framework is a powerful tool that should be standard protocol for investigating suitable management options for multiple sites. It is not a new concept, having being first formulated by Holling in 1978 (Holling 1978) and later advocated and developed by many others (e.g. Park 2004, Williams 2011). In particular, the development of the monitoring component of this framework appears to be an important addition (Lindenmayer and Likens 2009). However, adaptive management and monitoring have by no means become a standard tool for tackling conservation challenges (e.g. Brownstein et al. 2014, Williams and Brown 2014).

While study III by means of an adaptive management protocol allowed us to rule out mink as a salient factor limiting the breeding population of north Norwegian grebe, another approach is needed to investigate other sources of the reported population decline. The availability of breeding persistence data consisting of annual recording of a substantial number of breeding sites over a 22 year period allowed us to use an observational time series approach in Study IV. Although the monitoring was aimed at breeding success and not population trends, the scale of change in persistence suggested a large-scale change in factors affecting grebe on the breeding grounds or on their wintering grounds. The decline involved a decrease in both persistence and the number of grebe per active site over the period. None of the habitat variables explained the decline in persistence, however persistence was significantly greater with increasing number of pairs as measured at the start of monitoring in 1991. Similar declines have been recorded in adjacent populations of grebe in Scotland and Sweden without any pertinent factors being identified (Douhan 1998, Ewing et al. 2013, Pöysä et al. 2013).

Determining the reasons for decline is hampered by the lack of demographic data. Determination of standard methodology for counting has been developed (Stien unpub.) but remains to be tested over larger latitudinal gradients and with suitable sampling methodology. Future work should also focus on determining site faithfulness of breeding individuals to determine whether the decline is determined by mortality or movement to new breeding areas. As wintering grounds of individuals are unknown, tracking of individuals to their wintering grounds could also produce pertinent knowledge of factors

that could act negatively on survival during the winter season (e.g. Bogdanova et al. 2011, Delmore et al. 2012, Tranquilla et al. 2014).

4.3. Results and discussion for paper V

The general level of interest in mink control programs is surprisingly low, given that the number of small game hunters in the NAHA was more than 80, 000 in 2012 (SSB 2013a). The low response compromised the strengths of using a questionnaire approach to investigating collaborative efforts to control mink populations at a national scale. Having considered this limitation, there were some clear results that came out of the study and that can be used to determine the appropriate organisation of mink control programs and are relevant to the planning of invasive alien species programs in general.

The majority of mink hunters believed that mink is a conservation concern and particularly to ground nesting birds. Most were interested in conservation of non-red listed birds, including eider and other non-specified red listed species and is likely to reflect both the past and present Norwegian culture involving subsistence of coastal economies from down and egg collection of eiders (Soot-Ryen 1941) as well as the strong interest for grouse and ptarmigan hunting (Kaltenborn et al. 2012). Awareness campaigns for mink and generally for invasive alien species may well benefit therefore, from discovering and highlighting appealing species in areas where red listed species are targeted, in order to achieve conservation goals.

We found support for the hypothesis that mink hunters that received bounty caught more mink than those not receiving bounty, suggesting that introducing bounty as an incentive may result in an increase in mink catch. Evidence shows that well planned control programs that include bounty can be successful in invasive species control (Gosling and Baker 1989, McLeod et al. 2011, Newsome et al. 2014). The planning phase is crucial to a successful control campaign and, should take into account the appropriate spatial and temporal scale to prevent harvesting (Einarsson et al. 2006a, Bonesi and Palazon 2007, Bodey et al. 2009, Bryce et al. 2011, Pasko et al. 2014).

Contrary to expectations regarding the importance of sociality in motivating small game hunters (Andersen et al. 2008), and for hunting participation in general, we found that hunters that do not need to hunt near their home or together with friends and family caught on average more mink than hunters motivated by local hunting and social benefits. Many volunteer studies indicate that social benefits are important for being involved in and long-term participation in volunteer conservation efforts (e.g. Ryan et al. 2001, Asah and Blahna 2012) suggesting a positive relationship between volunteer retention and a sense of belonging (Selinske et al. 2015). Effective mink hunters may well therefore be 'loan wolves' but in common with established invasive alien species are unlikely to be able to cover the scale required to control mink in many areas. Thus, the long-term stability of initiated

control programs may be higher with an element of feedback that includes the whether the goal of large-scale control has been achieved. Mink hunters that were motivated by social interactions and local hunting caught significantly fewer mink. However, these hunters can also be highly relevant to successful control campaigns at landscape scale when organized into networks (e.g. Bryce et al. 2011).

The majority of mink hunters preferred governmental-led programs rather than citizen or landowner-initiated programs. Moreover, the hunters in our survey requested a more active role by the government for providing infrastructure, bounty, trapping equipment, organize landowner permissions to hunt, and to set up recruitment and education programs to increase participation in mink hunting. Favouring top down organisation may well be a direct result of the corporatist style of governance common to all Scandinavian countries and lack of conservation organisations and representation at the local level (Dryzek et al. 2002). It has also been seen to be crucial in achieving ecological and social objectives for invasive alien species management programs in Australia (Ford-Thompson et al. 2012). Given the low interest in mink removal and the history of low participation in previous programs, efforts should initially be concentrated around “conservation hotspots”, involving contract operators, bounty payments and awareness and recruitment programs.

5. Concluding remarks and recommendations

In this thesis I have used a variety of study methods at different spatial and temporal scales to explore population processes and management of 2 bird species of conservation concern, common eider and Slavonian grebe, focusing particularly on the role of nest predation in declining populations. I have demonstrated the role of proximate and ultimate causes of nest loss in eider at 2 declining colonies and highlighted the negative effects of human disturbance on nesting success and the potential role of disturbance in decline at one of the colonies. Furthermore, using the stepwise approach of elucidating proximate and ultimate factors has indicated management actions that are highly likely to improve nesting success and may also play a significant role in halting the decline of the eider colonies. I have demonstrated the usefulness of combining investigation of focal species of conservation and targeted invasive alien predator species in order to optimize management decisions when knowledge of both species is poor. I have further demonstrated the regional decline of Slavonian grebe at the northern range of its distribution. Finally, by investigating the level of mink hunting in Norway I have demonstrated that collaborative control programs that include volunteer mink hunters are best focused on hotspots of conservation concern. I view this thesis as an example of adaptive protocols for learning by doing. In contrast to common (mis)conceptions, I have demonstrated that this approach can be undertaken with relatively little funding and resource use and is simple in its design. Thus, conservation management in general, and game management particularly in Norway should be more willing to tackle management challenges using this adaptive protocol approach.

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