Management decisions and knowledge gaps: learning by doing in a case of a declining population of slavonian grebe Podiceps auritus

Authors: Jennifer Stien, and Rolf A. Ims
Source: Wildlife Biology, 21(1) : 44-50
Published By: Nordic Board for Wildlife Research
URL: https://doi.org/10.2981/wlb.00026

Wildlife Biology 21: 44–50, 2015
doi: 10.2981/wlb.00026

Subject Editor: Aurélien Besnard. Editor-in-Chief: Ilse Storch. Accepted 5 November 2014

Management decisions and knowledge gaps: learning by doing in a case of a declining population of Slavonian grebe *Podiceps auritus*

Jennifer Stien and Rolf A. Ims

J. Stien (jennifer.stien@uit.no) and R. A. Ims, Dept of Arctic and Marine biology, UiT- The Arctic Univ. of Norway, NO-9037, Tromsø, Norway

Species of national conservation concern require management action to reduce the threat of extinction. As part of its obligations to reduce national loss of biodiversity, the Norwegian authority for nature management (The Norwegian Environment Agency) published an action plan in 2010 for one of these species, the Slavonian grebe *Podiceps auritus*. The American mink *Neovison vison*, a non-native, invasive species with wide spread negative effects on native fauna, was highlighted as a major potential treat. We used an adaptive management approach that included management trials with the aim to assess whether mink predation is likely to be affecting grebe numbers significantly. We monitored mink activity, and put in place mink control measures at three of our seven study lakes. We then used 35 pairs of artificial nests, with one of each pair equipped with cameras, to measure predation at all seven lakes. The combined use of progressive experiments in an adaptive management/monitoring framework showed that mink activity was generally low with a mean activity at raft stations of between 0.41–1.22 per lake (n = 5), a range of zero to three excavations executed as a result of hunting (n = 3), and no incidences of mink nest predation (n = 35). Hence we conclude that mink is presently not likely to be a significant negative factor on grebe breeding success in the targeted lakes. We found a high nest predation rate by hooded crow with 18 of 21 identified predation events being identified to this species. Future effort should investigate non mink related threats to the Slavonian grebe such as the role of hooded crow in nest predation. This case study exemplifies the usefulness of the adaptive management/monitoring framework as a powerful means of testing hypotheses and to inform management, especially when knowledge of the focal system is poor.

All signatory countries to the Convention on Biodiversity (CBD) agreed to significantly reduce the rate of biodiversity loss by 2010 (Secretariat of the Convention on Biological Diversity 2010). Conservation measures to achieve this target include implementation of management to allow population growth of targeted species vulnerable to extinction and control of invasive species which can have a strong negative effect on vulnerable populations (Bonesi and Palazon 2007, Lentini et al. 2013). The Red List of species conservation (IUCN) is internationally recognised as the source for assessing the vulnerability of populations to extinction. It is the source which state nature management authorities use when compiling management strategies (action plans) to halt species biodiversity loss and therefore results in a large number of species which require some kind of management. However, for very many of these species it is unclear what kind of action is needed. An adaptive management framework is suitable for improving knowledge about system state as well as for assessing the relevance and effectiveness of potential management actions (Lindenmayer and Likens 2009, 2010, Williams 2011). Adaptive management may be carried out using an active or passive approach. The effects of multiple scenarios of perturbations on the response are tested in active management whereas the effect of a single perturbation is tested in passive adaptive management. Surprisingly, few studies utilise an adaptive management approach even though the idea has been around since the 1980s (Walters and Holling 1990, Williams 2011, Westgate et al. 2013).

Norway is a signatory to the CBD and as part of its obligations produced an action plan to increase the breeding population of Slavonian grebe *Podiceps auritus* (hereafter grebe) which was initially red listed in 2006 (Kålås et al. 2006) and currently has a small population with unknown trend. In Norway, Slavonian grebe breeds on inland water bodies between May and September (Cramp and Simmons 1977, Fjeldså 1973, 2004, but see Ulfvens 1988). Nests consist of a floating raft made from dead plant material situated in waterside vegetation. The action plan names 10 factors which are proposed to affect grebe negatively with all except one relating to factors at the species' breeding grounds (Fig. 1). However, direct evidence for these effects is lacking. It further expressed the need for increasing information about proposed, but untested, negative factors affecting the grebe at its breeding sites and implementing appropriate management (Direktoratet for Naturforvaltning 2009). We

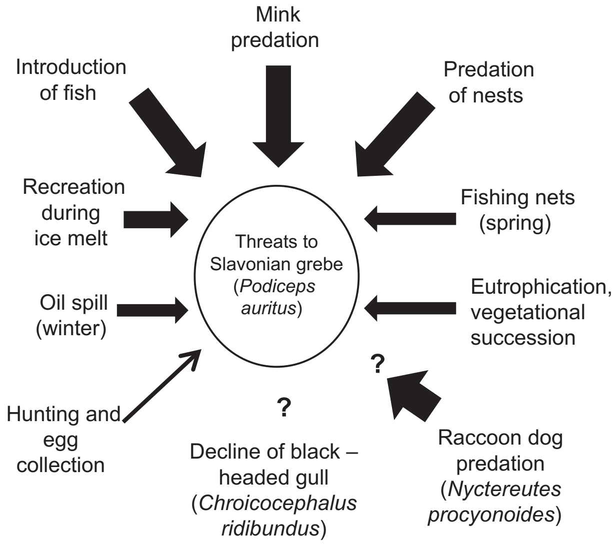

Figure 1. Study design showing factors perceived to have negative impacts on the Norwegian breeding population of grebe according to the Norwegian national action plan. Arrow thickness represents action plan assessed importance of the risk of a negative effect with thick arrows representing highest risk. Importance of risk is assessed as unknown for black – headed gull and potentially high for raccoon dog.

used an adaptive management framework to focus on one proposed high risk factor, the invasive American mink *Neovison vison*. Our target system was a core area for the grebe in northern Norway where the breeding population during the last decade has been subjected to a dramatic decline (Strann et al. 2014). We proceeded with management and monitoring trials in a sequential fashion in order to assess the usefulness of implementing mink removal as a management action to increase the grebe population in the target system. Each trial consisted of an experiment whose outcome informed the direction of each subsequent trial.

Material and methods

Experimental lakes and their grebes

Our focal *Podiceps auritus* conservation area, consisting of 7 lakes, is situated in Troms county, northern Norway which is an historical core area for the national population (Direktoratet for Naturforvaltning 2009). The grebe population in this area has been subject to a sharp decline in recent years (Fig. 2, Strann et al. 2014). The seven lakes are low lying (between 7 and 162 m above sea level with a mean area of 3.1 km^2, range 0.3–13.1 km^2) and oligotrophic with patches of shallow water vegetation consisting of water horsetail *Equisetum fluviatile* and common bog sedge *Carex limosa*. All lakes were relatively close to the sea (mean distance was 6.7 km, range 0.48–21.4 km). Lakes were in the same region and near each other so as to represent a single management area. They 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 lakes had connecting riparian ways of between 14.7 and 21.3 km. Estimates of number of breeding pairs of grebe per lake during the period 2001–2011 were obtained from Strann and Frivoll (2010) and Strann et al. (2010, 2012).

Management design and actions

Three sequential experiments were undertaken. The first 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 (Fig. 2). The second experiment proceeded in the same treatment and control lakes. Dogs were used to track mink on snow in early spring of 2011, while lakes were still frozen and before the arrival of the grebes. The third experiment focused on assessing the predation risk on grebe nests in the following breeding season in 2011.

Experiment 1. Mink removal and activity assessment with rafts and traps

Six mink raft stations consisting of a floating monitoring footprint plate and removable trap were deployed at each lake (Reynolds et al. 2004, Anonymous 2007, Bryce et al. 2011). Distance between each trap was 1 km allowing multiple detection possibilities within active mink territories at each checking period (Gerell 1970, Birks and Linn 1982, Reynolds et al. 2010). Stations were monitored every seven days to comply with the wildlife laws and to be a sufficient time interval to both remove individuals from a treatment area and record activity in control areas (Moore et al. 2003, Asakskogen 2010, Lambin pers. comm.). Stations were active during two periods during the ice-free season, with deployment between 26 May and 2 June as soon as ice melt began, until 21 July, and from 4 August until onset of first winter ice on October 20. This frequency of monitoring also ensured multiple sampling and trapping opportunities for mink in the study area. The start point for lake-raft deployment order was made by flipping a coin. All rafts were initially deployed in monitoring state (i.e. the tracking plate but not the trap was deployed). Traps were deployed at treatment site rafts only when tracks were registered on the monitoring plates. Lethal traps were used in order to further minimalize manpower required to check traps. Conibear 120 spring traps (<www.fangstmann.no/>) were used until 2 September, thereafter all treatment lakes monitored activity for 3 weeks while we waited for delivery of new traps. From 23 September Syningsfella (<www.syningfella.no/>) traps were used for the remaining four weeks until October 20.

As the objective of the study was to test the management action of mink removal, the first lake where mink activity was recorded became the first treatment lake (i.e. a lake where traps were deployed and activated in addition to monitoring). Thereafter the adjacent lake became a paired control lake where only monitoring of activity occurred. The next pair of lakes was similarly chosen whilst the remaining lake became a third treatment. Where activity had occurred, tracks were registered at each station and at treatment

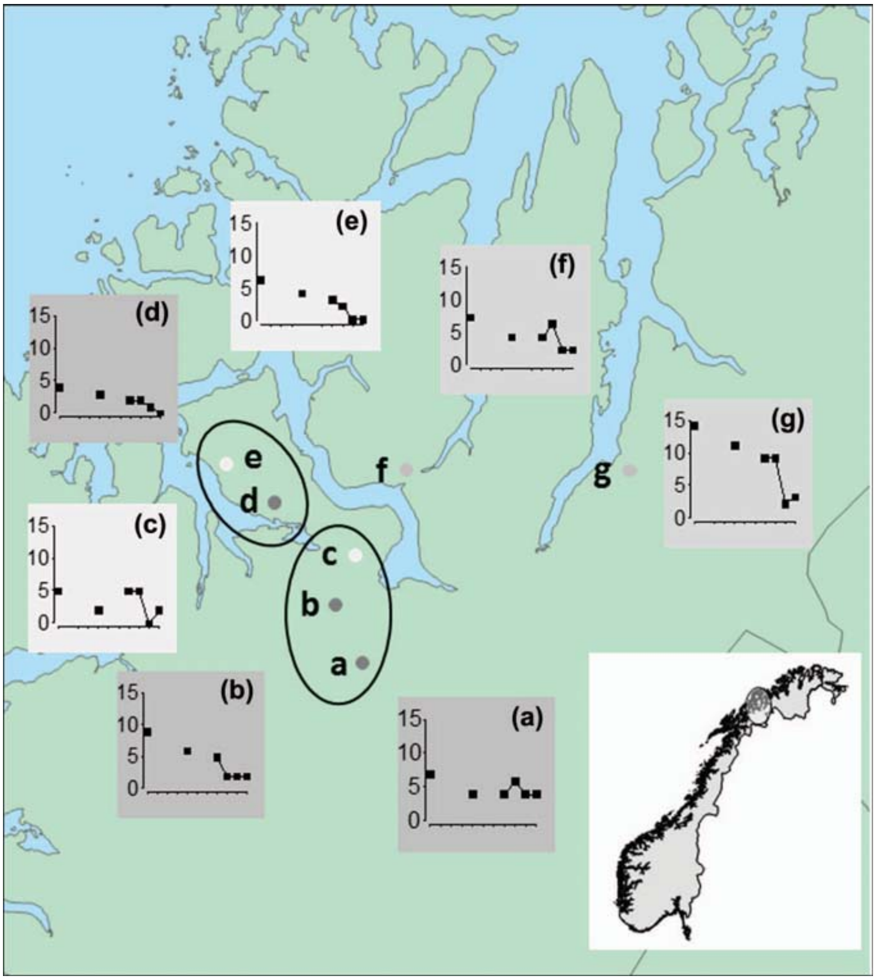

Figure 2. Site design and number of breeding pairs of grebe at study sites. Individual lakes are shown with dots. Colouring of dots relates to treatment (dark grey), control (pale grey) and additional (mid grey) lakes used for artificial nest experiment. Oval rings show the replicate treatment and control lake groups. Study lakes are (a) Langvatn, (b) Sagelvvatn, (c) Josefvatn, (d) Nordbyvatn and (e) Sandsvatn, (f) Laksvatn and (g) Øvervatn. Number of pairs is from 2001–2011, taken from Strann and Frivoll (2010) and Strann et al. (2010, 2012). Colouring of graph backgrounds is the same as for the dots. Inset map shows the study location area within Norway.

lakes a trap was deployed. At the next station round, traps and any caught mink were removed and footplates were returned. State of the trapping attempt (successful or not) was registered and resulting dead individuals were collected for later analysis. By-catch was also registered. After seven weeks of low activity and capture of only one individual all traps at treatment lakes were deployed permanently together with their respective tracking plates in order to maximize the chance of trapping transitory individuals between station rounds. Monitoring of mink at control lakes remained unchanged.

Experiment 2. Mink removal and activity assessment with snow tracking and dogs

The second experiment involved active searching and trapping (or tracking in control areas) of mink in the time window between establishment of mink territories and return of breeding birds. Active hunting works well when mink activity (and therefore the chance of mink entering a trap) is low (Nordström et al. 2003, Macleod, pers. comm.). Furthermore, the timing of mink removal provides temporary mink free areas during the breeding season and avoids disturbance of nesting birds (Craik 1995). A team of one biologist and two experienced hunters with two trained fox terrier dogs walked a transect line along each lake between 2 and 4 April 2011. Snow conditions were recorded during each transect as ‘good’ when a track could be easily identified and ‘poor’ when identification was not possible. Transects (mean = 6.12, range = 4.3–7.9 km) either circumvented small lakes or covered the area relevant to grebe nesting at larger lakes and was recorded as tracks on handheld GPS. Mink tracks crossing the transect line were logged with GPS as an index of activity. Active mink areas were located by dogs and also recorded on GPS. At treatment lakes an attempt was then made to remove the mink. Syningsfella traps were set in treatment areas where activity was registered but no mink was found. These were checked at the end of the week.

Experiment 3. Assessing predation risk with artificial nest

The third experiment followed the two trials of mink removals (experiment 1 and 2) and coincided with incubation and hatching of the earliest clutches of grebe. It was carried out at all seven lakes. It involved camera monitoring of artificial nests in grebe nesting habitat and was designed to assess egg predation risk due mink. We recorded by means of Garmin Map source habitat features which potentially could predict the variation in mink egg predation between lakes; specifically lake area and perimeter, altitude, distance from the coast and length of river tributary from the coast. Within the lakes we measured for each artificial nest the distance to closest incoming or outgoing waterways. Previous studies show that mink densities are higher in coastal than inland regions and also that mink mostly utilise waterways (Dunstone and Davies 1993).

Artificial nests of similar size and appearance to those built by grebe were made by binding lake vegetation around a small polystyrene raft. A quail egg was put in each nest to simulate an unattended nest during incubation phase. Nests were anchored using plastic coated gardening wire weighted by a stone. Five pairs of nests were deployed between 22–23 June and 14 July 2011 at each lake along a 1 km transect in grebe nesting habitat. Deviations occurred from 1 km when vegetation was patchy (mean 1.14 km, range 0.83–1.46 km). Transects were chosen to cross a major inflow or outflow maximizing the chance that mink would cross the area. Where several waterways existed, the transect line was chosen randomly. Mean distance between nest pairs was 284 m (range 151–469 m) and mean intra pair nest distance was 16 m (range 8–90 m). The first nest of each pair was deployed with a $14 \times 9 \times 6$ cm game camera. The camera was fastened to a thin stake 30 cm above the water line and approximately 3 m from the nest. Picture settings were set to motion in order to record movement at the nest. The second nest in each pair had no camera and functioned as a control for camera effects on nest survival (Richardson et al. 2009). We assumed that non camera monitored nests were subject to the same predator species and to the same predator behaviour as the camera monitored nests. The state of the nest, predated or not predated was recorded at the end of the period. Variation in predation rate between the seven lakes was analysed with simple logistic regression model in R with binomial distribution. The predictor variables area, altitude, river length and distance to nearest stream were entered singularly.

Ethical standards

All experiments comply with the current laws in which they were performed.

Results

Experiment 1

Activity of mink on the rafts was low in particular during spring and early summer when grebes are nesting and increased at all lakes during late summer (Fig. 3). Only two mink were captured from 259 trapping possibilities (i.e. at non malfunctioning stations on each of the treatment lakes) over a total of 89 trapping days, giving a frequency of 0.02 mink per trap day. There was one occurrence of by-catch of a waterfowl.

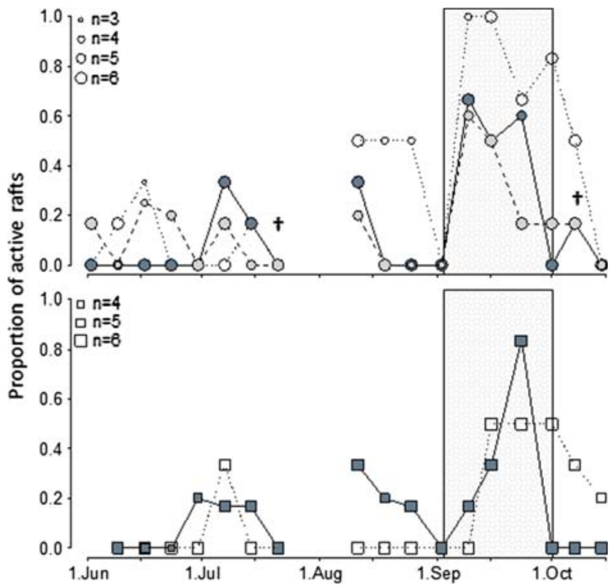

Figure 3. Mink activity shown as the proportion of mink raft stations with mink tracks recorded at seven day intervals in five lakes in Troms county between 26 May and 20 October 2010. Treatment lakes are shown by shaded symbols and control lakes by open symbols. Size of symbol shows the number of functioning rafts during trapping and recording of mink activity. Shaded rectangles depict periods of only monitoring in trapping lakes (see text for details). Lakes are Langvatn: dark grey circle, Sagelvvatn: light grey circle, and Josefvatn: open circle. Timing of capture of trapped mink at Langvatn is depicted by †; Sandsvatn: dark grey square, Nordbyvatn: open square.

Experiment 2

Snow conditions for tracking with dogs were good for all lakes except Sandsvatn where they were moderate as snow had begun to melt. Snow conditions for human visual tracking were poor at Sandsvatn. Mink activity assessed by means of dogs was low (Table 1). No mink were present in recently used holes and no individuals were caught by setting traps at places where mink activity was indicated by the dogs.

Experiment 3

Predation rate was high occurring at 56 of 70 artificial nests. At camera equipped nests, 27 out of 35 nests were predated. Predator species was detectable on 21 of these occurrences (Table 2). No mink egg predation was recorded. In addition, no pictures of mink were recorded in the vicinity of the nest.

Hooded crow *Corvus corone* was responsible for the majority of predation whilst jay *Garrulus glandarius* and common gull *Larus canus* were occasionally recorded. There was a significant heterogeneity among lakes in total predation rate (Fig. 4) with Nordbyvatn clearly deviating from the other lakes in terms of lower predation rate. Logistic regression on the proportion of surviving nests showed that none of the site scale predictors or distance to nearest stream were relevant ($p > 0.43$ for all models). Moreover, there was no difference in egg predation rates between camera or control nests (co-efficient estimate –0.35, $\pm$ 0.60, $p = 0.55$, DF = 68).

Table 1. Mink activity and trapping effort at study lakes between 2 and 4 April 2011. Transect length, snow conditions, number of times mink tracks crossed each transect line, number of mink holes found by dogs and number of times mink holes were excavated due to recent activity based on dog behaviour are shown. In addition the number of traps set inside the study area and in river outflows (outside of the study area), and number of days traps were active are also shown. Snow conditions were assessed as the same for both dog tracking and human visual tracking apart from at Sandsvatn where human visual tracking is shown in parenthesis.

Treatment /control	Lake	Date	Transect (km)	Snow conditions	Tracks (freq)	Holes (freq)	Excavations (freq)	Traps set (freq)	Traps set in outflow (freq)	Trap days (freq)	Mink killed (freq)
Treatment	Langvatn	2 April	7.9	Good	8	5	3	6	0	5	0
	Sagelvvatn	2 April	4.3	Good	0	0	0	0	2	4	0
	Nordbyvatn	3 April	5.1	Good	3	2	2	0	2	3	0
Control	Josefvatn	3 April		Good	0	0	–	–	–	–	–
	Sandsvatn	4 April		Moderate (poor)	4	0	–	–	–	–	–

Table 2. Artificial nest survival and frequency of egg predation by egg predator species for camera equipped nest and control nests (without cameras) between 22 and 23 June and 14 July 2011.

		Predator species at camera nests						
Lake	No. of camera equipped nests	Hooded crow	Jay	Common gull	Unknown	Survival at camera nests	No. of control nests	Control nests Survival
Josefvatn	5	4			1	0	5	0
Langvatn	5	3		1	1	0	5	0
Laksvatn	5	3			1	1	5	1
Nordbyvatn	5				1	4	5	4
Øvervatn	5	2	2			1	5	0
Sagelvvatn	5	5				0	5	0
Sandsvatn	5	1			2	2	5	1
Total	35	18	2	1	6	8	35	6

Discussion

The results are not consistent with the hypothesis that mink predation is the cause of decline in breeding numbers of Slavonian grebe in the region targeted by the present study. Monitoring of mink activity was generally low despite there being multiple possibilities for mink detection at each trapping and monitoring round. Activity was particularly low during the grebe nesting season. Furthermore, no predation event due to mink was recorded on the simulated grebe nests. Despite intensive trapping at 3 sites only two mink were caught. This is a low rate compared to other studies (Craik 2008, Harrington et al. 2008) even allowing for differences in capture methodology. Craik (2008) used non baited live traps, and caught an average of 3 mink year^{-1} per trap site during daily trapping sessions between June and the end of September at coastal sites. Harrington et al. (2008) used baited live traps and caught an average of 4.8 mink year^{-1} per trap site over a five-day trapping period in July and September at inland sites. These figures give a respective trapping rate per day of 0.73 and 0.64 in contrast to the trapping rate in this study of 0.02. We are not aware of any studies that compare live trapping capture rates with lethal traps or of studies that compare non baited traps to food baited. One study that compared the use of scent gland lure had mixed results (Moore et al. 2003).

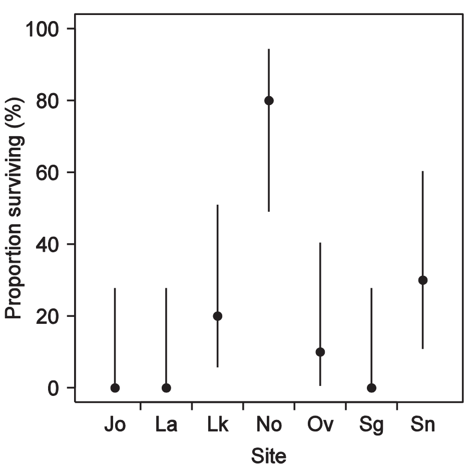

Figure 4. Survival estimates from logistic regression model with 95% CI for artificial nests at seven sites in Troms between 22 (Sandsvatn, Nordbyvatn, Josefvatn, Sagelvvatn) and 23 June (Langvatn, Øvervatn, Laksvatn) and 14 July (all sites) 2011.

Predation rate on artificial nests needs to be interpreted with care (Moore and Robinson 2004) and pertinent to the nesting behaviour of grebe may underestimate the risk of predation during intensive nest feeding activity during the first few days after hatching (Dillon et al. 2008). However, the high incidence rate of crow predation is in line with results from studies of crow diet (Coombs 1978, Zduniak 2006), studies of predation of natural nests on land during incubation (Stien et al. 2010) and hatching (Stien unpubl.), as well as studies of artificial nest predation in the Troms region (Pedersen et al. 2009, Klaussen et al. 2010). Furthermore, Summers et al. (2009) found a negative correlation between crow abundance and both grebe clutch survival and productivity. Crow and raven are included in the grebe action plan (Fig. 1) but without any empirical underpinning. Using an adaptive management approach, the next step would be to investigate crow predation of grebe nests. Camera monitoring at natural nests (Perkins et al. 2005, Richardson et al. 2009) could be used to test the hypothesis that nest predation is a key factor behind the decline of the grebe with crow being the most significant driver.

Our study indicates how an adaptive management framework can be implemented in order to increase knowledge for two species at opposing sides of conservation concern. Whereas grebe is vulnerable to extinction, American mink is seen as a key cause of local extinction of many ground nesting birds (Nordström et al. 2003, Bonesi and Palazon 2007). Our study provides previously lacking knowledge about mink activity in Norwegian lake systems as well the interaction between mink predation pressure and grebe nesting numbers and success. Thus, although mink is perceived as a cause of population decline for this species, implementing mink control as a management action aimed at preventing the decline of grebe in these areas is likely to be ineffective. Heterogeneity in predation pressure and predation species is seen in other studies of ground nesting birds (Chalfoun et al. 2002, Bolton et al. 2007, Stien et al. 2010) and is important to identify at sites targeted for management in order for the implemented management to be effective. Furthermore, utilising this progressive learning approach, which in our study involved three sequential experiments in the core area of grebe decline enabled us to rule out mink control as a management option within 1.5 years of initiation of the study. Thus the time lag between identification of need to act and appropriate action is minimalized. Finally, our study enabled us to identify crow nest predation as a more probable driver of grebe population declines in the study area.

Where several factors are proposed as being drivers of target species dynamics, active adaptive management is often more appropriate as multiple hypotheses are tested concurrently (according to an active management scheme; Williams 2011). However, there is seldom funding for this approach within nature management. This study shows that passive adaptive management as defined by Westgate et al. (2013) also has value within the nature management setting. Using the passive adaptive management approach, factors in the model would be tested sequentially.

Although countries have committed themselves to implement measures to reduce the loss of biodiversity, there will always be a lag in identification of species to be targeted and initiation of management. Poor knowledge of causes of declines hinders our ability to restore populations; adaptive management provides a framework for resolving this issue. It involves first making a conceptual model of potential drivers of population decline. These are then tested one at a time, improving our knowledge of their relative importance, and ultimately helping us focus conservation efforts (Lindenmayer and Likens 2010). It uses a framework which is simple to understand and highly relevant to the management goal for the conservation of the species. It is strange that the adaptive management approach has not been more widely adopted in conservation management (Westgate et al. 2013). We echo Westgate and Likens and advocate a call for a revival of this approach.

Acknowledgements – We thank Malangen-, Vestre Balsfjord-, Laksvatn-, Takvatn- and Sagelvatn- landowner associations and Statskog S.F. for permission to deploy mink traps on their land, and to Renathe Lohne for assistance in the field. Funding was provided by the Norwegian Environment Agency, Troms County Governor and the University of Tromsø.

References

Asakskogen, E. et al. 2010. Jakt og fangst av mink. – Norsk Jeger og Fiskeforbundet.

Anonymous 2007. The GCT mink raft. – Game Conservancy Trust.

Birks, J. and Linn, I. 1982. Studies of home range of the feral mink, *Mustela vison*. – Symp. Zool. Soc. Lond. 49: 231–257.

Bolton, M. et al. 2007. The impact of predator control on lapwing *Vanellus vanellus* breeding success on wet grassland nature reserves. – J. Appl. Ecol. 44: 534–544.

Bonesi, L. and Palazon, S. 2007. The American mink in Europe: status, impacts and control. – Biol. Conserv. 1344: 470–483.

Bryce, R. et al. 2011. Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management. – Biol. Conserv. 1441: 575–583.

Chalfoun, A. D. et al. 2002. Nest predators and fragmentation: a review and meta-analysis. – Conserv. Biol. 162: 306–318.

Coombs, F. 1978. The crows: a study of the corvids of Europe. – B. T. Batsford Ltd., UK.

Craik, J. 1995. Effects of North American mink on the breeding success of terns and smaller gulls in west Scotland. – Seabird 17: 3–11

Craik, J. 2008. Sex ratio in catches of American mink – how to catch the females. – J. Nat. Conserv. 16: 56–60.

Cramp, S. and Simmons, K. E. L. 1977. *Podiceps auritus* horned grebe. – In: Cramp, S. and Simmons, K. E. L. (eds). The birds of the Western Palearctic. Oxford Univ. Press.

Direktoratet for naturforvaltning 2009. Handlingsplan for horndykker *Podiceps auritus*. – Rapport 2009–7.

Dillon, I. A. et al. 2010. Provisioning of Slavonian grebe *Podiceps auritus* chicks at nests in Scotland. – Bird Study 574: 563–567.

Dunstone, N. and Davies, J. 1993. The mink. – Poyser.

Fjeldså, J. 1973. Territory and the regulation of poulation density and recruitment in the horned grebe *Podiceps auritus auritus* Boje, 1822. – Vidensk. Meddr Dansk Naturh. Foren 136: 117–189.

Fjeldså, J. 2004. The grebes: Podicipedidae, – Oxford Univ. Press.

Gerell, R. 1970. Home ranges and movements of the mink *Mustela vison* Shreber in southern Sweden. – Oikos 21: 160–173.

Harrington, L. A. et al. 2008. Estimating the relative abundance of American mink *Mustela vison* on lowland rivers: evaluation and comparison of two techniques. – Eur. J. Wildl. Res. 541: 79–87.

Klaussen, K. B. et al. 2010. Prevalence of nest predators in a sub-Arctic ecosystem. – Eur. J. Wildl. Res. 563: 221–232.

Kålås, J. A. et al. (eds) 2006. Norsk Rødliste 2006 – 2006 Norwegian Red List. – Artsdatabanken, Norway.

Lentini, P. E. et al. 2013. Effect of planning for connectivity on linear reserve networks. – Conserv. Biol. 27: 796–807.

Lindenmayer, D. B. and Likens, G. E. 2009. Adaptive monitoring: a new paradigm for long term research and monitoring. – Trends Ecol. Evol. 9: 482–486.

Lindenmayer, D. B. and Likens, G. E. 2010. Effective ecological monitoring. – Earthscan.

Moore, N. P. et al. 2003. Mink (*Mustela vison*) eradication to protect ground-nesting birds in the Western Isles, Scotland, United Kingdom. – N. Z. J. Zool. 30: 443–452.

Moore, R. P. and Robinson, W. D. 2004. Artificial bird nests, external validity, and bias in ecological field studies. – Ecology 856: 1562–1567.
Nordström, M. et al. 2003. Effects of feral mink removal on seabirds, waders and passerines on small islands in the Baltic Sea. – Biol. Conserv. 1093: 359–368.
Pedersen, Å. Ø. et al. 2009. Spatial and temporal patterns of artificial nest predation in mountain birch forests fragmented by spruce plantations. – Eur. J. Wildl. Res. 554: 371–384.
Perkins, A. J. et al. 2005. Use of time-lapse video cameras to determine causes of nest failure of Slavonian grebes *Podiceps auritus*. – Bird Study 52: 159–165.
Richardson, T. et al. 2009. Review and meta-analysis of camera effects on avian nest success. – J. Wildl. Manage. 732: 287–293.
Reynolds, J. et al. 2004. Development of population control strategies for mink *Mustela vison*, using floating rafts as monitors and trap sites. – Biol. Conserv. 1204: 533–543.
Reynolds, J. C. et al. 2010. Detectability of American mink using rafts to solicit field signs in a population control context. – J. Wildl. Manage. 74: 1601–1606.
Secretariat of the Convention on Biological Diversity 2010. – Global biodiversity Ourlook 3, Montréal. <www.cbd.int/doc/publications/gbo/gbo3-final-en.pdf>, last accessed 15.01.2014
Stien, J. et al. 2010. Nest predation in declining populations of common eiders *Somateria mollissima*: an experimental evaluation of the role of hooded crows *Corvus cornix*. – Wildl. Biol. 16: 123–134.
Strann, K. B. and Frivoll, V. 2010. Overvåking av hekkende horndykker i Troms 2009. – NINA Minirapport 290.
Strann, K. B. et al. 2010. Overvåking av hekkende horndykker i Troms 2010. – NINA Minirapport 323.
Strann, K. B. et al. 2012. Overvåking av hekkende horndykker i Troms 2011. – NINA Minirapport 374.
Strann, K. B. et al. 2014. Overvåking av hekkende horndykker i Troms 2013. – NINA Minirapport 485.
Summers, R. W. et al. 2009. Correlates of breeding success of horned grebes in Scotland. – Waterbirds 32: 265–275.
Ulfvens, J. 1988. Comparative breeding ecology of the horned grebe *Podiceps auritus* and the great crested grebe *Podiceps cristatus*: archipelago versus lake habitats. – Acta Zool. Fenn. 183: 1–75.
Walters, C. J. and Holling, C. S. 1990. Large scale management experiments and learning by doing. – Ecology 71: 2060–2068.
Westgate, M. J. et al. 2013. Adaptive management of biological systems: a review. – Biol. Conserv. 158: 128–139.
Williams, B. K. 2011. Adaptive management of natural resources: framework and issues. – J. Environ. Manage. 92: 1346–1352.
Zduniak, P. 2006. The prey of hooded crow *Corvus cornix* l. in wetland: study of damaged egg shells of birds. – Pol. J. Ecol. 543: 491–498.