LUTRA

Journal of the Dutch Mammal Society

Volume 68 – Number 1/2 – December 2025

Zoogdier vereniging

Volume *Deel* 68 – Number *Nummer* 1/2 – December 2025

Lutra is a scientific journal published by the Dutch Mammal Society (Zoogdiervereniging). The society is dedicated to the study and protection of native mammals in Europe. Lutra publishes peer-reviewed scientific papers on mammals across all disciplines, but tends to focus on ecology, biogeography, behaviour and morphology. Although exceptions are made in some cases, Lutra generally publishes articles on mammal species native to Europe, including marine mammals. Lutra publishes full articles as well as short notes which may include novel research methods or remarkable observations of mammals. In addition Lutra publishes book reviews. Lutra publishes in British English and, on request, in Dutch. Lutra is an open access journal and Lutra has no page charges. Lutra publishes one issue per year and Lutra is a member of Crossref.

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Art: Ed Hazebroek (p. 20, red fox; p. 40, European mink; p. 62, garden dormouse), Elias Nauta (p. 79, American mink)

ISSN 0024-7634

Carnivores: introductions and alien invasive species

This issue of Lutra has two articles on mink. One is on the feasibility of reintroducing the almost extinct European mink (*Mustela lutreola*). The authors (Zwartenkot et al.) come to the conclusion that the Netherlands has a considerable number of suitable areas for the reintroduction of the European mink. A few months ago Springer Nature published the volume on carnivores as part of the Handbook of European Mammals. It showed that, apart from Russia, the critically endangered European mink is now restricted to southwestern France, northern Spain, the deltas of the Danube and the Dniestr (Romania and Ukraine) and the Ukrainian Carpathians. Since the mid-19th Century its range has contracted by 90%. There is an introduced population on the island of Hiiuma (Estonia). This introduction succeeded, but two recent introductions in Germany (in Saarland and Steinhuder Meer) failed. This shows that we should not take the success of an introduction in the Netherlands for granted.

One of the drivers for the disappearance of the European mink is its displacement by the invasive, alien, American mink (*Neogale vison*), which brings us to the second article on mink in this volume of Lutra. La Haye traces the disappearance of feral American mink in the Netherlands to the closure of mink farms during the COVID-19 period. Apparently, the presence of free-living American mink was being sustained by constant escapes from mink farms. The disappearance of American mink in the wild offers a unique opportunity to reintroduce the highly endangered European mink in the Netherlands, which takes us back to the article by Zwartenkot et al.

The successful reintroduction of the otter (*Lutra lutra*) in the Netherlands shows that introduction of a semi-aquatic carnivore can succeed. This success also shows that water quality has improved, for example, with a reduction in pollution from organochlorine substances. On the other hand, there may be a risk of intraguild predation, which means that the well-established otters may kill European minks, which will be present in low numbers just after release.

Whereas the impact of pollution on otters seems to be resolved, this may not be the case for the garden dormouse (*Eliomys quercinus*), which just like the European mink, has lost a large part of its range. The cause of the disappearance is still not clear, but recently an article published in Germany suggested that pesticides may have a negative effect. Van Norren et al. provide a first indication of the exposure of the Dutch garden dormouse population to currently used pesticides. The Netherlands has only two populations of garden dormouse left and one is the result of an introduction. Both populations consist of not more than a few dozen animals.

The American mink is listed on the European Union List of Non-Native Species of Concern, as is the raccoon (*Procyon lotor*). This means that EU member states should at least try to control American mink and raccoon. Van den Berge et al. describe the settling of raccoons in Flanders. After decades of records of raccoons that must have been just local first-generation escapes, a raccoon population has been establishing itself in Flanders since 2014. Van den Berge et al. come to the conclusion that it is useless to start a full-scale eradication campaign, as this simply is not going to work, in view of the experiences in Germany.

Raccoons are considered to be a threat to indigenous breeding birds, as they raid bird nests. This could be a reason for local control of raccoons. To protect the nests of meadow birds the Province of Friesland had issued a licence to kill beech martens (*Martes foina*), but two months ago a court nullified the licence, stating that the Province should first improve the general conditions for the meadow birds. This would involve extensification of agriculture.

As all bat species are listed on Annex IV of the EU Habitats and Species Directive, compensation measures are often obligatory, in the cases of, say, development that will endanger their habitat or roosts. Boonman and Broer studied the occupancy of different types of bat boxes. The boxes were used by noctules (*Nyctalus noctula*) and brown long-eared bats (*Plecotus auritus*). It took five years until noctules started to use the bat boxes regularly and occupancy increased up to 25% after ten years. Noctules seem hesitant to accept new roost sites and/or need a lot of time to discover that bat boxes are suitable roost sites. Official guidelines suggest that placing bat boxes one year before trees are cut would give bats sufficient time to habituate to the new roost sites. However, this study shows that a lot more time may be needed.

The final paper in this volume is from Diekmann et al., who studied the effect of an agrivoltaic system on the presence of medium-sized mammals, including the raccoon, in Germany. Visit frequency and use intensity of the individual species hardly differed between the agrivoltaic site and the control site. However, the species observed are common and adaptable and it remains unclear how other more sensitive species might react.

Johan Thissen

Bat box occupancy by noctule and brown long-eared bat

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Abstract: Bats occupying 44 bat boxes in Park Randenbroek, Amersfoort, the Netherlands, were studied during a twelve-year period. The main aim was to determine how much time is needed before bats are using the boxes regularly and if exposition, mounting height and box type influence occupancy. During 16 autumn inspections, noctules (*Nyctalus noctula*) and brown long-eared bats (*Plecotus auritus*) were encountered in bat boxes 45 times (n=212) and 11 times (n=54) respectively. Large cylinder-shaped boxes were used more often by noctules and contained more individuals than small cylinder and flat boxes. Noctules preferred bat boxes with a southern exposition but no effect of mounting height was detected within our range of 3.7-7 m. Brown long-eared bats preferred small cylinder-shaped boxes over flat boxes. It took five years until noctules started to use the bat boxes regularly and occupancy increased to 25% after ten years. Noctules seem hesitant to accept new roost sites and /or need a lot of time to discover that bat boxes are suitable roost sites. Guidelines suggest that placing bat boxes one year before logging commences would give bats sufficient time to habituate to the new roost sites. Our study shows that a lot more time may be needed. Occupying new roost sites may be a group decision; this provides a possible further explanation for the long habituation period needed.

Keywords: bat box occupancy, noctule, *Nyctalus noctula*, brown long-eared bat, *Plecotus auritus*, exposition, mounting height, habituation.

Introduction

Bat boxes are often used to compensate for the loss of bat roost sites due to logging or forest maintenance. The underlying assumption is that bat boxes provide a suitable alternative for natural tree cavities that bats select for roosting. This is not always the case. Maternity roosts are rarely encountered in bat boxes and there is a general agreement that boxes cannot fully compensate for the loss of natural tree roost sites (Chambers et al. 2002, Zahn & Hammer 2016, Griffiths et al. 2017). But because bat boxes are commonly used by many bat species, it should be considered as a valuable tool to reduce at least some negative effects of logging.

There are dozens of different types of bat boxes commercially available, while at the same time bats have specific demands for their roost sites (Mering & Chambers 2014). Not every bat box has the temperature, size and shape that bats prefer. The way bat boxes are installed can also influence occupancy. A large number of bat boxes is more effective than a small (<10) number of boxes (Rueegger 2016). More exposure to sunlight will increase the temperature of the bat box and close to the ground bats may not have sufficient ground clearance. Bat boxes installed at low (<2.5 m) height are also vulnerable to

Table 1. Characteristics of 44 bat boxes that were monitored. Exposition = compass direction from the centre of the bat box towards the space directly in front of the box.

Box type	Number	Height	Exposition			
		Average (min-max)	N	E	S	W
Schwegler 1FF	19	5.3 (3.8-6.9)	6	7	3	3
Schwegler 2FN	18	4.9 (3.6-7.0)	7	5	5	1
Schwegler 1FS	5	5.4 (3.7-6.6)	2		1	2
Schwegler 1FW	2	6.1 (6.0-6.1)	1		1	

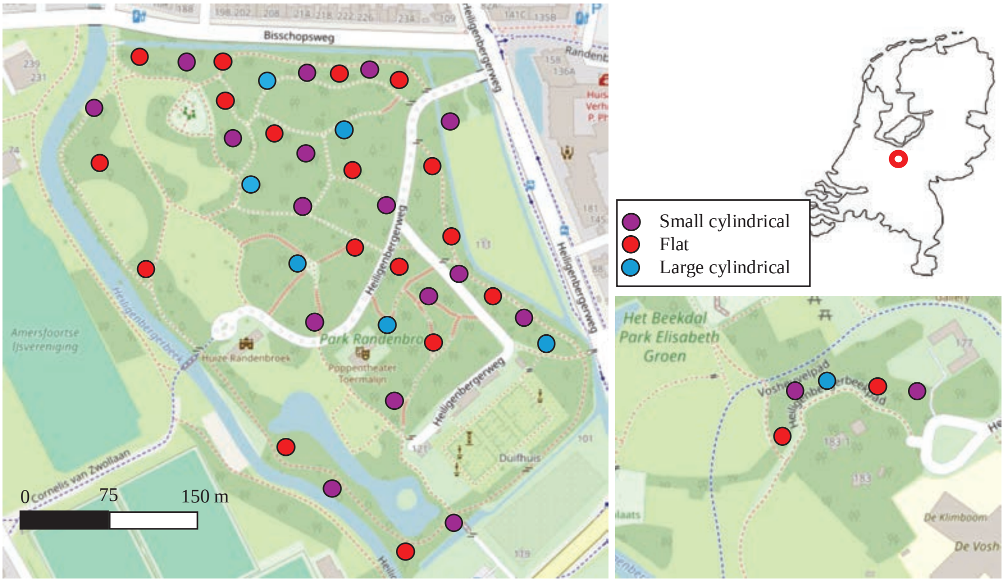

Figure 1. Location of the different bat box types within the study area Park Randenbroek.

vandalism. Although some guidelines for bat boxes exist (Korsten 2012, Rueegger 2016, BIJ-12 2017), a lot of expert judgement is still needed, as guidelines often simply state that the bat boxes should have the same quality as the original natural roost sites that they compensate for. This is unfortunate because much information about bat box occupancy has already been gathered by monitoring. By analysing monitoring data from 44 bat boxes during a twelve-year period, we aim to contribute to the knowledge of bat box occupancy by two forest dwelling bat species that can be used for more detailed guidelines. Our study focused on the following questions:

- How much time does it take before bat boxes are used by bats?
- Is there an effect of box type, mounting height and exposition on bat box occupancy?

Method

Study location

Monitoring of bat boxes took place in Park Randenbroek, a former estate situated in the city of Amersfoort, the Netherlands (Figure 1). The city park consists mainly of beech (*Fagus sylvatica*) forest in the last stage of succession. During maintenance of the park in 2013, dozens of mature trees were logged. One large (>20 animals) and 2-3 smaller (<10 animals) noctule (*Nyctalus noctula*) roosts were present in the park in the years prior to 2013 (Boonman & Brekelmans 2012).

Bat boxes

In March 2012 47 bat boxes were put up on mature trees in the park. Three bat boxes disappeared over the years and were removed from the analysis. Four different bat boxes were used: Schwegler 1FF (flat box with wide opening), 2FN (small cylinder), 1FS (large cylinder), FW (large cylinder with thick wall). We reduced the width of the entrance of the 2FN boxes to 15 mm to prevent frequent use of the boxes by birds. All four bat box types are composed of a mixture of cement and wood. Except for Schwegler 1FW, all bat box types were installed at a range of mounting heights and exposition (Table 1).

1FF and 2FN boxes were alternately placed in the tree lanes to ensure that in every section of the park all bat box type were present in equal quantity (Figure 1).

Monitoring

Between 2012 and 2024 all bat boxes were checked 16 times in total. 1FF boxes can be checked without opening them. All other bat box types were opened and the number of bats present was counted. Identification was done according to Dietz & von Helversen (2004). Bats were not handled unless this was necessary for safely closing the lit after the inspection. Unused bird nests and large amounts of guano were removed if this was feasible without disturbing the bats. During the first five years the bat boxes were checked 2-3 times a year in spring, summer and autumn. From 2015 checks were no longer part of a funded monitoring programme and were continued by volunteers of the Bat Group Amersfoort. Due to the very low bat box occupancy in spring and summer, checks were done once a year during autumn from 2017 onwards. During the covid years bat boxes were not checked. During every visit, all bat boxes were checked.

Statistical analysis

Before 2016 and during spring, the number of bats occupying the boxes was too low for quantitative analysis. Therefore, we only used data collected after 2016 during autumn inspections. Data from both Schwegler 1FS and 1FW boxes were combined as box type 'large cylinder', since both boxes have approximately the same size and shape. To determine the influence of bat box type, height and exposition on occupancy by bats we used a binomial GLMM from the lme4 package of R (R development core team). The observations (presence or absence of bats in a box) are not fully independent because the same bat boxes were checked during every inspection. When bats use a bat box, their scent and droppings are left behind. This may influence the chance of future occupancy. We incorporated this dependency in the model by using bat box ID as random effect. We tested an effect of exposition by looking at the deviation from north. Differences between north and south in occupancy can be detected by using this method.

Results

During the first four years of monitoring, the boxes were used only incidentally by bats. Only during a few inspections bats were encountered and only a few bat droppings were found. After four years, occupancy started to increase for both noctule and brown long-eared bat (*Plecotus auritus*) (Figure 2). Occupancy increased to 25% during 2022 and 2023 for noctules. Noctules were encountered in bat boxes 45 times (*n*=212). Brown long-eared bats were seen in boxes 11 times (*n*=54). A Nathusius' pipistrelle (*Pipistrellus nathusii)* was seen once. Never were two species seen in the same box at the same time.

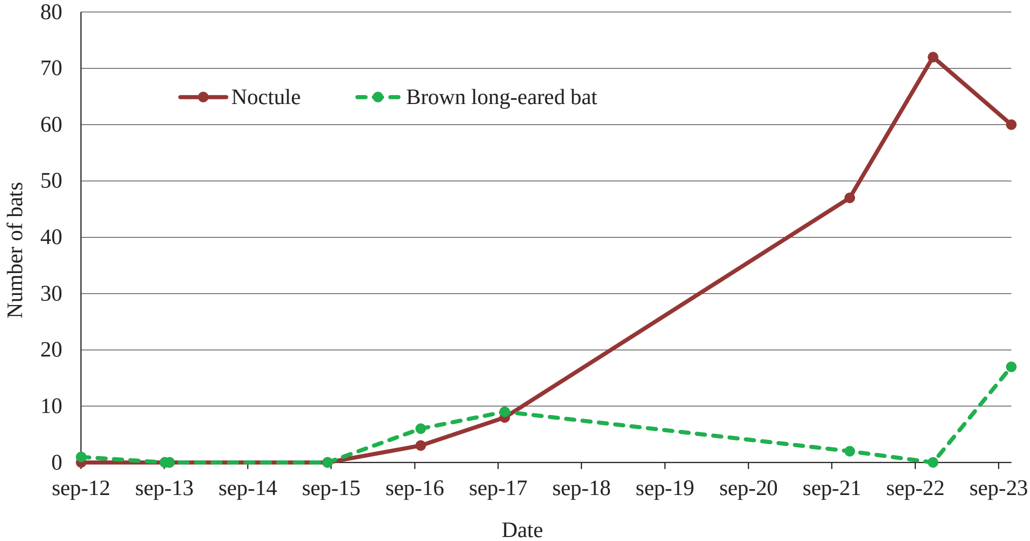

Figure 2. Total number of noctules and brown long-eared bats encountered in 44 bat boxes in autumn between 2012 and 2023. During covid-years no inspections were carried out.

Influence of box type, height and exposition

Box type and exposition had a significant effect on bat box occupancy by noctules (Table 2). Approximately one third of the larger cylindrical bat boxes were used by noctules, compared to one fifth and one sixth for flat boxes and small cylindrical boxes respectively. The larger cylindrical bat boxes contained on average 3.4 noctules per inspection compared to 0.4 and 0.6 for flat boxes and small cylindrical boxes (Figures 3 and 4). The largest number of noctules encountered in a bat box (Schwegler FS) was 19. Occupancy of the bat boxes by noctules decreased when boxes were more exposed to the north, hence showing a preference for a southern exposition. Bat box height did not affect occupancy.

Table 2. Results of the binomial GLMM, comparing the occupancy of the large cylindrical (LC) boxes by noctules with the occupancy of small cylindrical (SC) and flat boxes (F). The influence of height above ground and exposition (deviation from north) on occupancy is also shown. * significant

Covariate	Z-value	Probability (*P*)
Box type LC: SC	-2.0	0.047 *
Box type LC: F	-1.3	0.19
Deviation from north	2.3	0.023 *
Height above ground	0.5	0.64

Brown long-eared bats were encountered in bat boxes 11 times (n=54). This number is too low for an extended statistical analysis. Brown long-eared bats were found more often in small cylindrical boxes (n=8) than in flat boxes (n=2). This difference is significantly different from a distribution over both box types that would occur if there was no preference for box type (Chi-squared test χ^2= 3.9, df=1, P<0.05). Only once were brown long-eared bats found in large cylindrical boxes. The highest number of brown long-eared bats found in a single bat box was 17 (Figure 5).

Discussion

Bats often use distinctly different roost sites during each part of their yearly life cycle. In winter bats select roost sites that are suitable for hibernation, such as caves with a high

Figure 3. Two noctules in large cylinder-shaped bat box (Schwegler 1FS). *Photo: Erik Broer.*

humidity and a constant temperature. In summer the same bat species may be found in warm and dry roost sites that are more suitable for raising young. Although noctules use trees as roost sites all year round, a shift in roost preference throughout the year is also likely in this species. In winter, larger roost sites may be preferred as larger group sizes are generally recorded in this time of the year. It is assumed that increasing group size facilitates enduring low ambient temperatures (Gaisler et al. 1979). More subtle changes can also be expected during the year as reproductive females spend less time in torpor compared to post-lactating females (Dzal & Brigham 2013). Our study focused on bat box occupancy in autumn. The results of this study may not be applicable for occupancy in spring and summer when bats may have different preferences. Although the number of bat boxes and checks is relatively low, our wide range in mounting height and exposition that is evenly spread over the study area is rarely shown in other studies. Merging the results from different studies that each have limited variation in covariates is helpful to obtain a bigger sample size, but will certainly result in unbalanced data. In many areas for example, only a single bat box model is used. Bats can also be more common in a specific area, leading to a high bat box occupancy. In this case it will be impossible to tell whether high occupancy is the result of a preference for this box type or a preference for this area.

In our study, noctules showed a preference for large boxes with a cylindrical shape. These boxes are three times more expensive than small cylindrical boxes. Occupancy in larger boxes is twice as good and the average number of noctules encountered in these boxes is six times larger. Therefore, buying larger boxes seems to be the best choice for this species, and well worth the extra costs. Brown long-eared bats preferred the smaller cylindrical boxes over flat boxes in our study. Flat boxes have a wide opening at the bottom and may not provide the best microclimate in autumn. It should be noted that the cylindri-

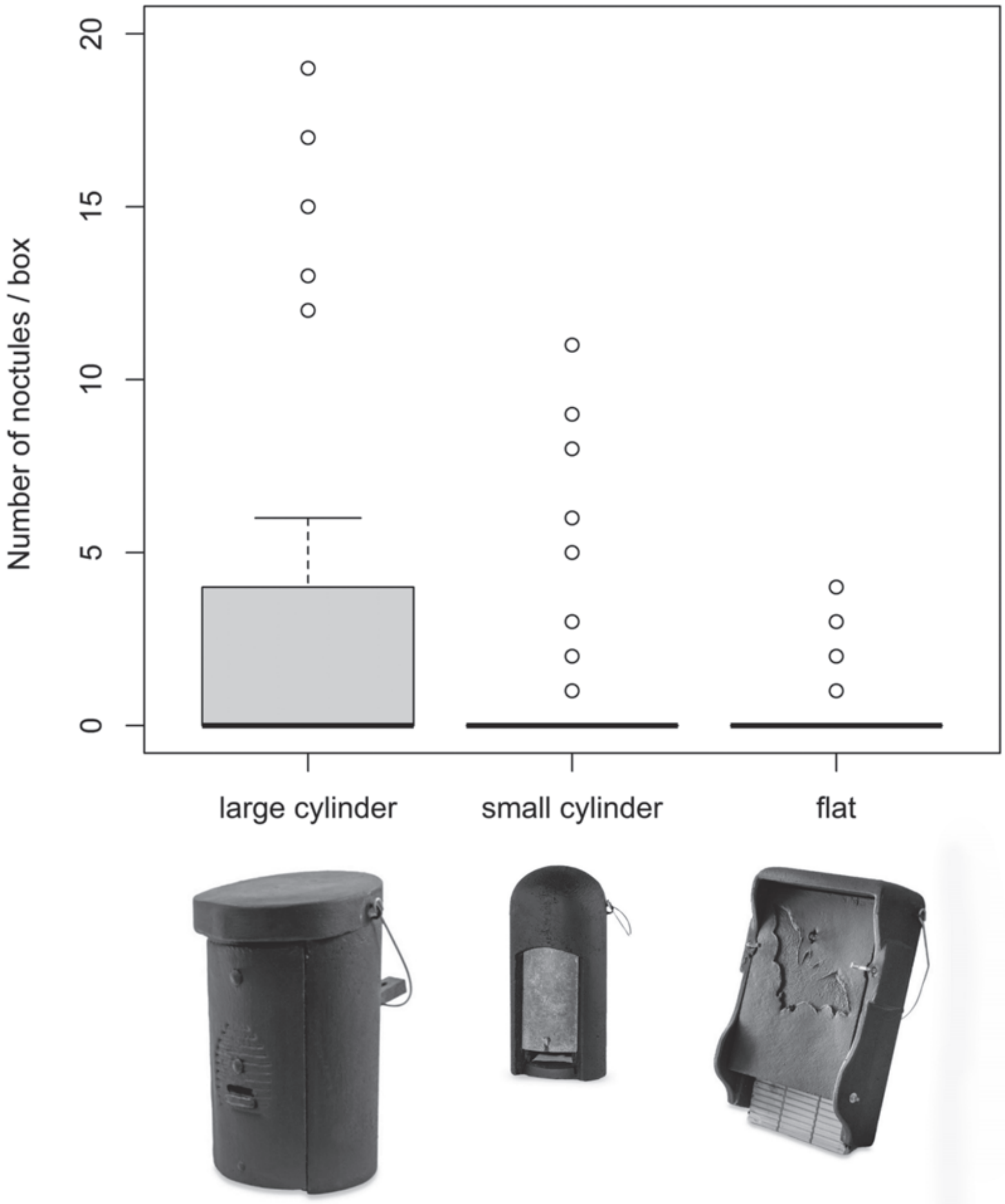

Figure 4. Boxplot of the number of noctules present in three different box types: large cylindrical (left), small cylindrical (middle) and flat box (right).

cal bat boxes were cleaned by us during the study period. Without cleaning, the boxes can fill up with bat droppings, which may negatively affect occupancy. Pschonny et al. (2022) found a preference for flat boxes in common pipistrelle (*Pipistrellus pipistrellus*), a preference also seen in Nathusius' pipistrelle (Korsten 2012). These species-specific differences are probably related to the natural roost sites that bats prefer. Noctules prefer tree cavi-

Figure 5. Seventeen brown long-eared bats occupying a small cylinder-shaped bat box (Schwegler 2FN). *Photo: Erik Broer.*

ties constructed by woodpeckers (Boonman 2000) while Nathusius' pipistrelles often use the space behind tree bark as roost site (Dietz et al. 2009).

The mounting height of the boxes did not affect occupancy within our range of 3.6 to 7 m. Other studies showed a weak positive effect of mounting height on occupancy in common pipistrelle and Natterer's bat (*Myotis nattereri*) (Pschonny et al. 2022). The mounting height of their boxes was substantially lower (2-4 m) than the boxes in our study. We expect, therefore, that below 3.6 m there may indeed be an effect of mounting height on occupancy in noctules, which need sufficient ground clearance during emergence. Our study showed a preference for boxes with a southern exposition in noctules. Although the boxes are located under the forest canopy, the boxes facing south may still receive a bit more sunlight, making them slightly warmer than boxes with a northern exposition. Kerth et al. (2001) also found a preference for warmer boxes in Bechstein's bats (*Myotis bechsteinii*) in autumn. Printz et al. (2021) on the other hand found a preference for NW exposed boxes (that only receive sunlight at the end of the day) in noctules in spring. Bats may prefer to spend more time in torpor in spring compared to autumn.

One year after the bat boxes were mounted, dozens of mature trees were logged. This was done under surveillance of ecologists. Each tree was carefully inspected from a cherry picker and cavities were inspected with an endoscope camera before felling to make sure no bat was killed. The logging reduced the availability of potential and known roost sites for noctules (Boonman & Brekelmans 2012) and led to disturbance during the autumn of 2013, but not to high mortality in the local population. Printz et al. (2021) suggested that a lower density of roosting opportunities may lead to an increase in bat box occupancy. We could, however, not detect an increase in bat box occupancy during the first four years after the trees had been logged. The logging led to a gradual increase in forest undergrowth over the twelve-year period (personal observation), which may have led to an increased food availability for brown long-eared bats. Contrary to long-eared bats, noctules are aerial hawkers, adapted to forage in open areas (Norberg & Rayner 1987, Mackie & Racey 2007), and do not forage in dense forest undergrowth. It took five years before noctules began to use the boxes in our study regularly. It is unlikely that this reflects changes in local population size. Noctules populations may have increased, such as observed in the western part of the Netherlands (Mostert & Bekker 2024), but due to the noctule's low reproduction rate such an increase is taking place relatively slow (Zukalova et al. 2022), i.e. has been occurring over several decades and not just over the last ten years. We think that it is more likely that noctules need a lot of time to discover that bat boxes are suitable roost sites and / or that they are hesitant to accept new roost sites. Guidelines (BIJ12 2017) suggest that placing bat boxes one year before logging commences would give bats sufficient time to habituate to the new roost sites. Our study shows that much more time may be needed. Griffiths et al. (2017) and Pschonny et al. (2022) also showed that occupancy of bat boxes depends on the age of the boxes, an effect that may last for up to ten years. This even exceeds the life expectancy of most individuals. A possible explanation for the extensive amount of time needed to accept new roost sites is that bats are social animals that prefer to roost in a colony or in close vicinity of conspecifics. Fleichmann & Kerth (2014) showed that in brown long-eared bats group decision making about communal roost were made unanimously. Pre-dawn swarming behaviour in tree-dwelling bats is believed to drive the collective selection of new roosts (Zelenka et al. 2020). Accepting new roost sites as a group will take more time because more animals need to agree with the decision. As a consequence of this extensive habituation time, monitoring bat boxes may not be suitable for detecting population trends in noctules. Other bat species can be less selective. There are examples of flat bat boxes that are frequently used by Nathusius' pipistrelles within one year after installation (personal communication J. Boshamer, own observation). Additionally, bats that are already familiar with bat boxes may accept new bat boxes more quickly (Bergmann et al. 2025).

Acknowledgements: We would like to thank members of the Bat Group Amersfoort who played an indispensable role in checking the bat boxes. Thanks also to the anonymous reviewers for improving our manuscript.

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Samenvatting

Het gebruik van vleermuiskasten door de rosse vleermuis en de gewone grootoorvleermuis

Vleermuiskasten worden veel gebruikt om de effecten van het kappen van bomen op boombewonende vleermuizen te verminderen. Het type kast en de wijze waarop ze worden opge-

hangen kan gevolgen hebben voor de kans dat ze door vleermuizen gebruikt worden. De richtlijnen die hiervoor zijn opgesteld, zijn niet altijd concreet, wat veel ruimte overlaat voor een eigen interpretatie. Dit is jammer, omdat inmiddels veel gegevens beschikbaar zijn over het gebruik van vleermuiskasten. We hebben in deze studie het najaarsgebruik geanalyseerd van 44 houtbetonnen vleermuiskasten in Park Randenbroek in Amersfoort over een periode van twaalf jaar. Daartoe hebben we bepaald hoelang het duurde voordat de kasten in gebruik genomen werden en of er een effect op het gebruik is van het type kast, de hoogte waarop deze was opgehangen en de expositie. De vleermuiskasten werden opgehangen één jaar voordat de kap van tientallen volwassen bomen plaatsvond. De eerste vier jaar werden vleermuizen en vleermuiskeutels slechts incidenteel aangetroffen. Het gebruik van de kasten door rosse vleermuizen (*Nyctalus noctula*) in de herfst begon na vijf jaar toe te nemen en bedroeg een kwart na tien jaar. Het lijkt erop dat het erg lang duurde voordat rosse vleermuizen ontdekten dat de vleermuiskasten als verblijfplaats gebruikt kunnen worden en /of dat de soort erg terughoudend is in het accepteren van nieuwe verblijfplaatsen. Mogelijk speelt hierbij een rol dat het in gebruik nemen van een nieuwe verblijfplaats een groepsbeslissing is in plaats van een individuele keuze. Grote cilindervormige kasten werden het meest gebruikt door rosse vleermuizen en bevatte ook het hoogste aantal dieren. Rosse vleermuizen bleken een voorkeur te hebben voor kasten met een zuidelijke expositie. Binnen de range van 3,6-7 m was er geen significant effect van hoogte boven de grond op het gebruik. Gewone grootoorvleermuizen (*Plecotus auritus*) gebruikten kleine cilindervormige kasten vaker dan platte kasten. We hopen dat vleermuiskasten effectiever ingezet kunnen worden met behulp van deze informatie.

Received: 19 September 2025
Accepted: 29 October 2025

Case study on the use of a high-mounted agrivoltaic system by mammals – results of a camera trapping survey over a one-year period

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Abstract: Agrivoltaic systems involve the dual use of land for agriculture and solar energy generation and, as such, in the context of renewable energy expansion can reduce the competition for land between photovoltaic expansion and food production. So far, however, there is a lack of knowledge on the effects of this combined land use type upon biodiversity. To gain initial insights, we used camera traps to investigate the habitat use of a small-scale, high-mounted agrivoltaic trial plot and an adjacent control plot (identical land use, but without solar panels) by large and medium-sized mammals over a one-year period in northern Germany. Our results showed no major differences between the plots regarding the number of recorded mammal species and their activity. Visit frequency and use intensity of the individual species hardly differed between the agrivoltaic plot and the control plot. However, the species found were common and adaptable (e.g. brown hare, roe deer, red fox, raccoon) and it remains unclear how other, less adaptable species might react. In addition, it is questionable whether these results can be transferred to larger agrivoltaic facilities or other systems (e.g. tracking modules) and further research on this question will be needed.

Keywords: biodiversity, species richness, solar energy, renewables, camera traps.

Introduction

Driven by technological innovation and political support, the expansion of renewable energies, particularly solar energy, is progressing at an increasing rate (Nijsse et al. 2023). This development will lead to a stronger competition for land, especially between energy generation and food production (Nonhebel 2005, van de Ven et al. 2021). To contain this conflict, agrivoltaic systems could be a possible solution, as they are a dual land-use, combining energy generation with solar panels and agricultural production on the same land (Dinesh & Pearce 2016, Widmer et al. 2024). At the same time, the adaptation of agrivoltaic systems will – similar to conventional ground-mounted photovoltaic systems – lead to landscape changes due to visual impacts and changes in land use and openness (Sirnik et al. 2024). At present there is a lack of knowledge about the environmental effects of agrivoltaic systems on biodiversity (Gomez-Casanovas et al. 2023). It is assumed that some species may avoid agrivoltaic systems due to the UV-light emitted by supporting structures and that larger mammals could be affected by reduced movement ability (Schwarz & Ziv 2024). To our knowledge there are no existing field studies on the effects of agrivoltaic facilities on mammals. The few existing studies on mammal use of conventional solar parks emphasize that security

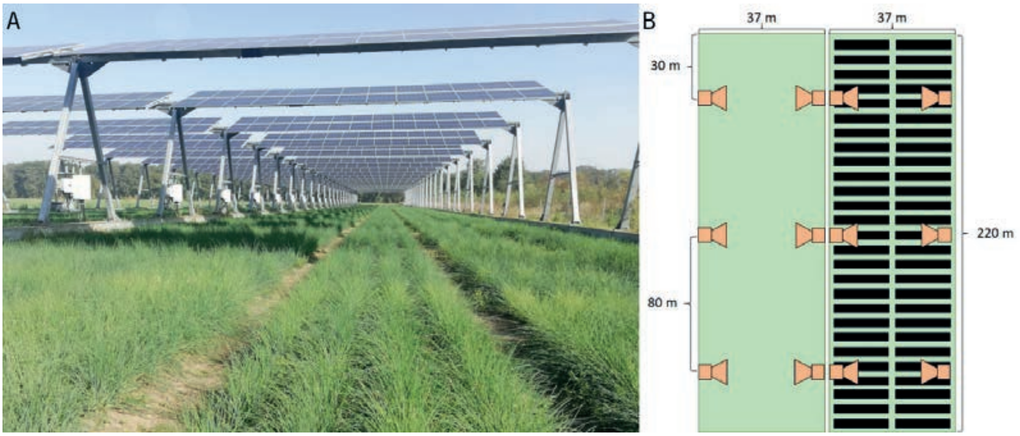

Figure 1. A. The agrivoltaic includes 24 rows of south-facing solar modules at a height of approx. 6 m. B. Both the agrivoltaic plot and the control plot had a size of 37 x 220 m and were surveyed with six CTs each (three at the east and three at the west side of each plot).

fences, typically installed around these facilities, can pose movement barriers, especially to large mammals, while small- and medium-sized mammals usually can pass through ground clearances under or gaps within the fences (Bennun et al. 2021, Sawyer et al. 2022). However, these findings are not transferable to agrivoltaic facilities since these are usually not fenced and may therefore have no or limited barrier effects on larger mammals. Furthermore, agrivoltaic facilities also differ considerably from regular solar parks in terms of land utilization: While the vegetation in conventional solar parks generally comprises extensive grassland, agrivoltaic focuses on agricultural use which may also affect their attractiveness for mammals. Overall, it is unclear whether mammals use or avoid these novel structures in our landscapes. To gain initial insights into this topic, we conducted what we believe is a first case study over a one-year period at a small-scale agrivoltaic facility in northern Germany. Our aim was to determine which large and medium-sized mammals use this agrivoltaic system and if there are any signs of avoidance behaviour or differences in visit frequency or use intensity in comparison to an adjacent control plot with identical land use, but without solar panels.

Materials and Methods

Study sites

Our research was conducted on an agrivoltaic trial plot built in 2022 in the north-east of Lower Saxony (district Lüchow-Dannenberg, northern Germany; coordinates: 53°00’13.5”N 11°10’26.5”E). The area is characterized by agriculture (arable land and grassland) with a forest in the north and structured by hedges and tree rows. The agrivoltaic trial plot has a size of approx. 0.8 ha (37 x 220 m) with 24 rows of high-mounted, south facing solar panels at a height of approx. 6 metres (Figure 1A). We investigated large and medium-sized mammals on two sample plots: the agrivoltaic plot itself (Agrivoltaic) and a directly adjacent plot without solar panels (Control). Both plots were of the same size and were sown with chives (*Allium schoenoprasum*) in September 2022. The chives on both plots were cultivated as a perennial crop, meaning intensive farming with frequent harvesting (about every 20 days during the vegetation period) and disturbance for maintenance (fertilizer or pesticide use several times a month in the growing season).

Survey methods

Mammals were recorded with camera traps (CTs) (Dörr SnapShot Mini Black 12 MP HD, Dörr GmbH, Neu-Ulm, Germany). We used six CTs per plot (Figure 1B) which triggered a Passive Infra-Red (PIR) sensor (Welbourne et al. 2016). The CTs were set to take a series of three images within three seconds per trigger, increasing the chance of identifying the species. After a delay of ten seconds the next trigger was possible. The CTs were active continuously from 17 January 2023 to 23 January 2024 (372 days).

On both plots, CTs were positioned facing the crop (Figure 1B) with three CTs installed on the western and three on the eastern side of the plots. The distance between these CTs was 80 m and a distance of approx. 30 m was kept from the northern and southern edges of the plots. As the detection range of the installed CT models is around 10–15 m, a major part of the area between the CTs (width of the plots approx. 37 m each) was covered. The CTs were mounted on the AGRIVOLTAIC sub-construction and on metal poles on CONTROL at a height of 70 cm with an inclination angle of approx. 5° towards the ground.

Data preparation and analysis

After the surveys, all information (date, time, location of CT, detected species) of images with large and medium-sized mammals was transferred to a table for further analysis. We only considered images of wild mammal species with at least the size of a red squirrel (*Sciurus vulgaris*). One table row was created for each detection (consisting of a series of three images, see CT settings). Based on this, the following values were calculated as indicators for the habitat use of mammals for the individual CTs, for the two plots (joint consideration of the six CTs used per plot) and overall (joint consideration of all CTs used): the number of detected species, number of detections and percentage of days (out of 372 days in total) with detection of the respective species. For all calculations, one camera day was defined as 24-h period beginning and ending at 12 a.m. (cf. Zitzmann & Reich 2022), including the entire dawn, dusk and night time, which are the main activity periods for most of the mammal species. Data was analyzed descriptively, as we only conducted investigations on two plots (AGRIVOLTAIC, CONTROL) for our case study and because the six CTs used per plot were nested within the same site and thus were not independent from each other.

Results

All CTs were continuously active during the investigation period (372 days per CT, 2232 camera days per plot and 4464 camera days in total). Overall, 3639 detections of mammals were made, distributed equally between AGRIVOLTAIC and CONTROL (Table 1). For more than 98% of the detections the species (or at least genus) could be identified. For 60 detections (1.6%) determination was not possible due to the poor image quality. Overall, six mammal species were detected: five of them on AGRIVOLTAIC and six on CONTROL. More than 60% of all detections were of the brown hare (*Lepus europaeus*) (Table 1). Roe deer (*Capreolus capreolus*; 24.6%) and red fox (*Vulpes vulpes*; 9.4%) also accounted for higher proportions. In contrast, raccoon (*Procyon lotor)*, martens (*Martes* spec; see$_1$ in Table 1) and especially wild boar (*Sus scrofa*) accounted for much fewer detections.

Of the detected species, the brown hare used both AGRIVOLTAIC (53% of days, with *n*=372 days) and CONTROL (62% of days) most regularly (Table 2, considering all CTs per plot). Roe deer and red fox were also regularly (23–31% of days) detected on both plots. The visit frequency on AGRIVOLTAIC and CONTROL were comparable for each of these three most common species. Raccoon and *Martes* spec. were only detected on a few days (max.

Table 1. Number of detected species and number of detections on both plots (with n=6 CTs per plot) and in total (n=12 CTs) during the survey period of 372 days.

	Agrivoltaic	Control	Total
European hare *Lepus europaeus*	1051	1205	2256 (62%)
Roe deer *Capreolus capreolus*	515	380	895 (24.6%)
Red fox *Vulpes vulpes*	156	186	342 (9.4%)
Raccoon *Procyon lotor*	33	21	54 (1.5%)
Genus *Martes*[1]	25	6	31 (0.9%)
Wild boar *Sus scrofa*	–	1	1 (0.03%)
Not determinable	40	20	60 (1.6%)
No. of species	5	6	6
No. of detections	1820	1819	3639

[1] The majority of images with representatives of genus *Martes* could only be identified to genus level, as characteristics for reliable species identification were rarely recognizable. Therefore, all detections with representatives of the genus *Martes* were considered together in the analyses. In some of the pictures, however, the stone marten (*Martes foina*) could be identified with certainty, while in none of the pictures could the pine marten (*Martes martes*) be reliably identified.

Table 2. Percentage of days with detection of the species during the investigation period (372 days) by at least one of the CTs used per plot (with n=6 CTs) und in total (n=12 CTs); Values in Brackets = No. of CTs with detection of the particular species.

	Agrivoltaic	Control	Total
Brown hare	53.0 (6)	61.8 (6)	72.0 (12)
Roe deer	22.6 (6)	29.6 (6)	40.3 (12)
Red fox	23.9 (6)	31.2 (6)	44.1 (12)
Raccoon	5.9 (6)	3.0 (5)	8.6 (11)
Genus *Martes*[1]	5.9 (5)	1.6 (4)	7.5 (9)
Wild boar	–	0.3 (1)	0.3 (1)

[1] See Table 1

6%) but also on both plots while wild boar was detected on just a single day on Control.

At CT-level (n=6 CTs per plot), mean and median values as well as standard errors regarding the number of species and number of detections per CT hardly differed between Agrivoltaic and Control (Table 3), but the range of values in numbers of detections was larger on the Agrivoltaic. Species-specific visit frequency and use intensity of the three most common species within the two plots hardly differed (Table 3), but especially for the brown hare, the range of values was larger on the Agrivoltaic, indicating big differences between individual CTs in this plot.

Discussion

The number of mammal species detected and overall mammal activity of Agrivoltaic and Control were comparable. Species-specific results showed that the three most common species (brown hare, roe deer, red fox) were recorded on slightly more total days on Control, but visit frequency of these species on CT level hardly differed between Agrivoltaic and Control, showing no clear difference in habitat utilization between both plots. However, all species detected are quite adaptable and also regularly occur in human settlements (stone marten, raccoon, red fox) or intensively used agricultural landscapes (hare, roe deer) (Hewison et al. 2001, Bateman & Fleming 2012, Santilli & Galardi 2016). Thus, they are used to anthropogenic structures and an avoidance behaviour towards the agrivoltaic system was therefore not to be expected, especially as studies from ground-mounted solar parks already confirmed habi-

Table 3. Mean values (±SE), median values (±SE), minimum and maximum of the measured variables per CT on the two surveyed plots (with n=6 CTs per plot).

	Agrivoltaic	Control
No. of species		
Mean	4.8 ± 0.2	4.7 ± 0.2
Median	5.0 ± 0.7	5.0 ± 0.8
Min - Max	4 - 5	4 - 5
No. of detections		
Mean	303.3 ± 49.9	303.2 ± 28.5
Median	319.0 ± 12.5	319.0 ± 9.5
Min - Max	138 - 512	155 - 370
Visit frequency[1]		
Brown hare		
Mean	20.9 ± 2.6	21.6 ± 1.7
Median	22.3 ± 2.8	20.0 ± 2.3
Min - Max	10.8 - 29.0	16.9 - 27.4
Roe deer		
Mean	9.1 ±1.5	9.0 ± 1.3
Median	9.4 ± 2.1	8.9 ± 2.0
Min - Max	4.6 - 13.4	4.6 - 14.2
Red fox		
Mean	5.6 ± 0.7	7.2 ± 0.9
Median	5.4 ± 1.5	7.8 ± 1.7
Min - Max	3.5 - 8.6	4.0 - 10.2
Use intensity[2]		
Brown hare		
Mean	175.2 ± 28.0	200.8 ± 19.0
Median	169.0 ± 9.4	216.5 ± 7.7
Min - Max	64 - 284	103 - 242
Roe deer		
Mean	85.8 ± 18.1	63.3 ± 12.5
Median	71.0 ± 7.5	65.0 ± 6.3
Min - Max	42 - 163	25 - 96
Red fox		
Mean	26.0 ± 3.7	31.0 ± 4.0
Median	27.0 ± 3.4	33.0 ± 3.6
Min - Max	13 - 39	16 - 44

[1] Visit frequency=Proportion of days [%] with presence of the particular species per CT
[2] Use intensity=No. of detections per CT

tat use, e.g. by hares and red fox (Herden et al. 2009, Montag et al. 2016).

Nevertheless, it cannot be ruled out that other mammal species than those detected by us now avoid the area due to the establishment of the agrivoltaic facility, or that the habitat use of the detected species considerably decreased, as we did not perform a before-after-comparison. Furthermore, it is conceivable that Agrivoltaic also influenced Control, as they were directly adjacent. Animal movement was thus not independent between both plots as individuals might have been deterred by the agrivoltaic system and thereby also be detected less frequently on Control compared to regular agricultural fields in the surrounding area. Overall, however, no strict avoidance of the investigated agrivoltaic system was observed, but regular use of the common species was detected.

One advantage of agrivoltaic facilities over traditional ground-mounted photovoltaics for mammal species is clearly the lack of fencing. This allows mammals of all sizes to use or cross the area without barriers, which is shown in our study by the frequent presence of roe deer. In contrast, the fencing of ground-mounted photovoltaics can lead to habitat loss and fragmentation, especially for large mammals (Sawyer et al. 2022). On the other hand, ground-mounted photovoltaics are relatively undisturbed due to their mostly extensive grassland use and consist of various microhabitats (e.g. locations under solar panels, between panel rows and edge areas; cf. Zitzmann et al. 2024). In contrast, (intensive) agricultural use on agrivoltaic facilities leads to more frequent disturbance and a more homogeneous habitat structure, which reduces the availability of food and cover, potentially making agrivoltaic systems generally less attractive for some species. The attractiveness of agrivoltaic facilities certainly also depends on the landscape context. For hares at least, the intensity of land use is less important for habitat use than the habitat diversity in the landscape (e.g. different crops and structural diversity) (Santilli & Galardi 2016). This matches the high presence of brown hare in our study on both Agrivoltaic and Control, despite intensive agriculture, and might indicate that especially small-scale facilities could possibly be of minor conflict potential for this species.

Finally, it must be emphasized that our study focused on a small-scale agrivoltaic facility. Thus, there were no real central areas, but it actually consisted exclusively of edge zones. Therefore, our results are not transferable to large-scale agrivoltaic facilities where central areas might be avoided by mammals. Furthermore, the type of agricultural land use (the type of crop along with the required management and its intensity), as well as the distance to other habitats such as forests will certainly have a strong influence on their attractiveness for and their use by mammal species (cf. Månsson et al. 2021). Accordingly, there is a need for further research on the effects of agrivoltaic on mammals and also on other species groups, especially at larger facilities, in different landscapes, with different crops (e.g. legumes or orchards) and different technical systems (e.g. vertical modules or tracking systems).

Acknowledgements: We thank R. Steinicke GmbH, on whose land and agrivoltaic system field work for this research took place. We also thank M. Graas for his support during the field work. We are also grateful to two anonymous reviewers for their helpful comments on an earlier version of the manuscript and to Nicholas Parrott of TextualHealing.eu for his English language editing assistance.

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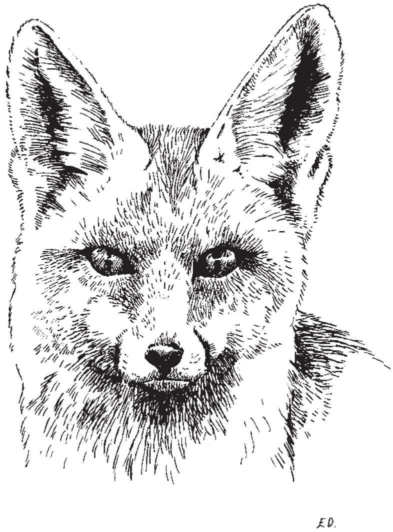
Samenvatting

Case study over het gebruik van een agrivoltaïsche opstelling door zoogdieren – resultaten van een cameravalonderzoek over een periode van één jaar

Agrivoltaïsche systemen omvatten het dubbel gebruik van land voor zowel landbouw als voor de opwekking van zonne-energie. Hierdoor kunnen ze de concurrentie om land tussen de uitbreiding van fotovoltaïsche installaties en voedselproductie verminderen in het kader van de groei van hernieuwbare energie. Tot nu toe is er echter weinig bekend over de effecten van dit type landgebruik op biodiversiteit. Om eerste inzichten te verkrijgen, hebben we cameravallen ingezet om gedurende een periode van één jaar op een locatie in Noord-Duitsland het gebruik door middelgrote en grote zoogdieren van een kleinschalige agrivoltaïsche proefopstelling met verhoogd (6 m) geplaatste zonnepanelen een aangrenzend controleperceel (met identiek landgebruik, maar zonder zonnepanelen) te onderzoeken. Onze resultaten toonden geen grote verschillen tussen beide percelen wat betreft het aantal waargenomen zoogdiersoorten en hun activiteit. Ook de bezoekfrequentie en de gebruiksintensiteit van de individuele soorten verschilden nauwelijks tussen het agrivoltaïsche en het controleperceel. De waargenomen soorten, zoals haas, ree, vos en wasbeer, zijn echter algemeen en flexibel in hun aanpassing, waardoor het onduidelijk blijft hoe andere, meer kritische soorten zouden reageren. Bovendien is het onzeker in hoeverre deze resultaten overdraagbaar zijn naar grotere agrivoltaïsche installaties of andere systemen (zoals meedraaiende zonnepanelen). Verder onderzoek is daarom nodig.

Received: 28 March 2025
Accepted: 18 July 2025


E.D.

The rise of the raccoon (*Procyon lotor*) in Flanders, Belgium: chronicle of a predicted evolution

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Abstract: As a non-native species, the raccoon (*Procyon lotor*) has been present in the wild in Europe for about a century, and the expectation seemed justified that it would establish itself also in Belgium. A meticulous recording and interpretation of all possible raccoon sightings in Flanders (northern Belgium) over the past 30 years shows that this was not the case during the first two decades of this period. In contrast, a clear increase in the number of registrations has occurred in the last decade. Necropsy of collected roadkills also shows that reproduction in the wild is occurring throughout the region, a finding confirmed by the increasing use of camera traps. The raccoon has been included in the Union List of Non-Native Species of Concern since 2016 through the EU Regulation, and addressing its further population development in Europe is currently the subject of international consultations. This study aims to provide a documented baseline for Flanders on the raccoon population status, while also making some considerations regarding the need for systematic intervention.

Keywords: raccoon, wasbeer, *Procyon lotor*, invasive species, management, Flanders, Belgium.

Introduction

The issue of wild-living non-native species is now a standard concern in efforts to preserve biodiversity. Alongside ecological impacts, there are often additional worries, such as the risk of zoonotic disease transmission or damage to property and economic interests. These risks are influenced by whether the species is invasive and the extent of its population growth in new environments.

Within this broader context, non-native predatory mammals, such as raccoon (*Procyon lotor*), raccoon dog (*Nyctereutes procyonoides*) and American mink (*Neogale vison*), have received attention for many years in Flanders, the northern part of Belgium. Their potential population development has always been followed with suspicion or curiosity, within the nature sector but often also in a broader public context, with regular press coverage. All three of the species mentioned have become established in various European countries and the fear or expectation that these species would soon settle in Flanders too seemed justified for several years. However, to date, this appears to be the case only recently with raccoons, while the presence of American mink and raccoon dogs in Flanders still remains limited to isolated sightings so far, according to the carnivore database of the Flemish Research Institute for Nature and Forest (INBO).

In Flanders, raccoons are covered by the Species Decree (2009), partim alien species. Introduction into the wild was therefore already prohibited. Since August 2016, EU Regulation No. 1143/2014 has also prohibited the private keeping or trading of raccoons. These restrictions were added to the implicit ban that already

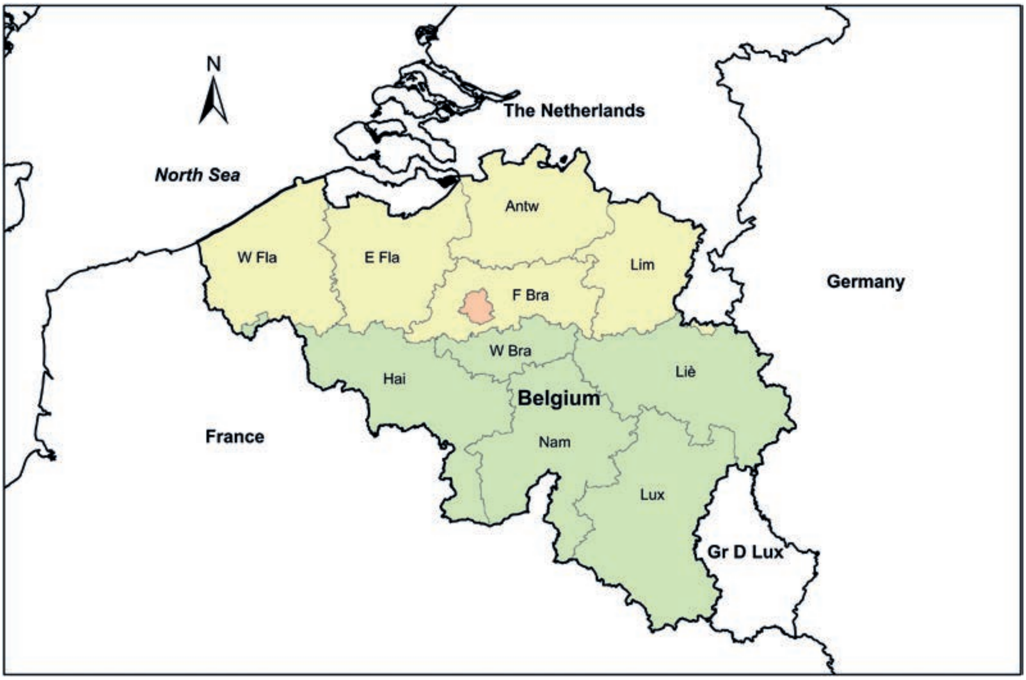

Figure 1. Location of Flanders in federal Belgium, with an indication of the regions (yellow: Flanders, green: Wallonia, orange: Brussels) and the provinces (W Fla: West Flanders, E Fla: East Flanders, Ant: Antwerp, F Bra: Flemish Brabant, Lim: Limburg, Hai: Hainaut, W Bra: Walloon Brabant, Nam: Namur, Liè: Liège, Lux: Luxembourg).

applied on the basis of the Belgian 'Positive List of Mammals', which does not include raccoons.

This paper aims to synthesize validated raccoon data to present an overview of both the recent past and the current population status of this species in Flanders. It does not aspire to provide a comprehensive risk analysis or practical management recommendations for raccoon populations. First and foremost, this overview can serve as a reference point for monitoring future developments. However, a number of critical considerations inevitably arise, with more questions being raised than definitive answers can be given at present.

Materials and methods

Study area

Flanders, the northern region of Belgium (Figure 1), covers approximately 13,500 square kilometres with an altitude ranging from 0 to 288 m. The temperate maritime climate is characterized by a mean annual precipitation of 780 mm and a mean annual temperature of 10.1 °C. The landscape is predominantly flat, with a network of rivers and canals shaping its hydrology. Flanders is among the most densely populated and industrialized regions in Europe (Jaeger et al. 2011). Forest coverage is only 10%, and urban expansion, intensive agriculture, and infrastructure development have significantly altered the natural environment (Statbel 2020). Native carnivore species that are widespread in the area include red fox (*Vulpes vulpes*), Western polecat (*Mustela putorius*), stoat (*Mustela erminea*) and weasel (*Mustela nivalis*), while badger (*Meles meles*), otter (*Lutra lutra*) and pine marten (*Martes martes*) are irregularly distributed (Van Den Berge & Gouwy 2021, Van Den Berge et al. 2021). Since 2018, the wolf (*Canis lupus*) has also settled in the area with (at least) one territory (Van Den Berge 2018, Van der Veken et al. 2021).

Surveying method

In 1998 a network of volunteers and professionals (such as forest rangers) was established at INBO, as part of the research on carnivores, especially mustelids (martens). This 'Marternetwerk' (Marten Network) aimed at centralizing all possible information about the presence of carnivores in Flanders into a specific carnivore database. Unlike neighbouring countries, Flanders does not have hunting statistics for most predators, as hunting these species has been closed for years. However, many carnivore species are often victims of traffic accidents, and such animals collected by the network provide reliable information about their presence and, through necropsies, insight into population parameters such as reproductive status, age distribution, and health. This helps distinguish between established populations and wandering individuals, which is especially important for rare, declining or emerging species. From the moment the Marten Network became operational, explicit attention was also drawn to the exotic predator species that could potentially be present in Flanders. After all, these species can lead to species confusion in the case of superficial assessment or damaged carcasses (e.g. badger versus raccoon or raccoon dog), while at the same time, *avant la lettre*, the problem of alien species was explicitly raised (Van Den Berge 1998).

In 2008, the online platform Waarnemingen.be became available through the private nature organisation Natuurpunt, which gradually took over the collection of sightings, while the collection of dead specimens became a complementary activity of INBO. The most robust data comes from collected dead animals, but all other sightings (live animals, tracks, etc.) from all kinds of sources (broader than waarnemingen.be, e.g. also from captures, press articles) are also integrated in the INBO-carnivore database, with careful attention to avoid double-counting. Strict reliability criteria are applied when assessing observations that are not linked to concrete evidence (photographs, collected dead specimens, etc.), e.g. via a cautious, exploratory questioning of observers. The data retained is divided into four categories, namely 'certain', 'very probable', 'possible' and 'undetermined'. Recently, the widespread use of camera traps by volunteer nature researchers provided a strong new tool for detecting highly elusive species, such as most mammalian predators, including raccoons.

Necropsy

Prior to necropsy, the collected specimens are frozen in an ultra-freezer (-80 °C) for 8 days for safety reasons with regard to possible parasitic infections. After thawing, a number of standard biometric measurements are recorded externally, including the total weight and the weight after removal of all internal organs. In some cases, the latter allows for a better comparison between specimens, as animals collected as roadkill are often no longer intact and no 'total' weight can be recorded. A general physiognomic assessment is also recorded, such as possible lactation in females, any coat lesions or an obvious juvenile stage. Evidence or strong indications of reproduction in the wild are considered very important in the context of a possible population development.

During the internal autopsy, in addition to taking various tissue samples (e.g. for genetic testing) and analyzing the stomach contents, attention is focused primarily on a number of characteristics relating to reproductive status and age. In females, the uterus is assessed for weight and any signs of pregnancy (embryos, placental scars). In males, the baculum is dissected, measured and weighed, and referenced to the age classification according to

Table 1. Overview of necropsy results from the collected raccoons with a view to determining possible reproduction in the wild – see text. (Information on origin or date in brackets may not be entirely precise and is subject to reserve).

Location	Date	Sex	Wt	We	HBL	Cra	Mol3cl	Mol2cl	Can	UtW	Gest	BacW	BacL
Denderleeuw	15/10/2000	M	-1	-1	-1	-1	-1	-1	-1			-1	-1
Peer	29/09/2001	M	4126	-1	57.5	-1	-1	-1	-1			1.608	91.9
Erpe-Mere	20/05/2002	F	3848	-1	54.5	2	2	2	3	1.04	NBP		
Willebroek	20/09/2002	F	6342	5304	56.5	-1	-1	-1	-1	3.1	NP		
Harre	3/08/2010	M	5234	-1	55.7	-1	-1	-1	-1			1.936	84.3
Laakdal	26/01/2011	M	-1	4776	-1	-1	3	2	3			2.844	95.6
Bertem	30/08/2012	M	4472	3824	55.5	3	2	2	3			2.732	95.9
Houston (USA)	1/2014	M	3446	3056	56	2	2	1	1			2.497	95.8
Dilbeek	2/2015	F	4202	3440	54	3	2	2	3	6.02	BP		
Dilbeek	6/2015	M	4118	3432	52.7	1	1	1	1			1.327	81.1
Dilbeek	6/2015	F	4912	4622	53.3	3	2	2	3	1.79	BP		
Hoeilaart	29/07/2015	M	2122	1758	46	1	1	1	1			0.31	50.4
Lille	23/09/2015	F	3010	2316	47.2	2	1	1	1	0.54	NBP		
Tervuren	30/07/2016	M	-1	3416	56	-1	2	2	2			1.938	90.5
Maasmechelen	17/10/2016	M	6664	5614	58.7	3	2	2	3			3.583	91.1
Lede	24/10/2016	M	6122	4958	58.5	3	3	2	3			3.232	92.5
Zottegem	19/02/2017	M	8120	6812	60.8	-1	3	2	3			5.162	99.1
Hasselt	18/01/2018	F	7646	5836	59	3	2	2	3	9.06	BP		
Hasselt	2/11/2018	M	7950	4578	57	-1	2	2	3			1.702	85.9
Duffel	11/02/2019	M	6048	5368	58	2	2	2	3			2.756	102
Maasmechelen	10/03/2019	M	5896	4530	54	-1	2	2	1			1.795	92
Voeren	13/11/2019	M	5732	4966	38	-1	1	1	1			1.238	80.7
Halen	6/02/2020	M	6418	4984	60	2	2	1	2			2.3	94.3
Herselt	18/02/2020	F	4910	4360	51.5	-1	2		3	7.47	P		
Oudsbergen	22/10/2020	M	4636	3670	53	-1	1	1	1			0.99	74.1
Schoten	21/07/2021	M	4078	3538	52.7	2	2	2	2			2.09	95.1
Bierbeek	(1/09/2021)	M	4234	3976	59	2	2	2	3			2.177	98.8
Westerlo	20/10/2021	M	4592	3760	54	1	1	1	1			0.946	77.7
Hasselt	19/06/2022	M	4010	3400	57.7	2	2	2	3			1.977	89.8
(Peer)	(1/08/2022)	M	4876	4400	-1	3	2	2	3			3.168	99.9
Diest	18/12/2022	F	4920	4920	57	-1	1	1	2	-1	-1		
Hasselt	5/10/2023	M	-1	3580	48.7	2	2	1	1			0.66	68.9

Schwery et al. (2011). For both sexes, the skull is inspected for closure of the cranial sutures and characteristics of the teeth, more specifically the degree of wear of the molars and the degree of closure of the canine root apical foramen (Grau et al. 1970), on the basis of which a distinction between juvenile and adult animals can be made with reasonable certainty.

Table 1 provides the following information:

- Origin, Date: reference of the necropsied specimen
- Sex: gender, M/F
- Wt: total weight in grams
- We: eviscerated weight (i.e. without entrails) in grams
- HBL: head-body length (i.e. from tip of nose to anus) in centimetres
- Cra: closure of the sutures of the bones of the skull (cranium); 1 = none, 2 = partial, 3 = complete closure
- Mol3cl: degree of wear on the molars in three classes; 1 = none, 2 = limited, 3 = advanced

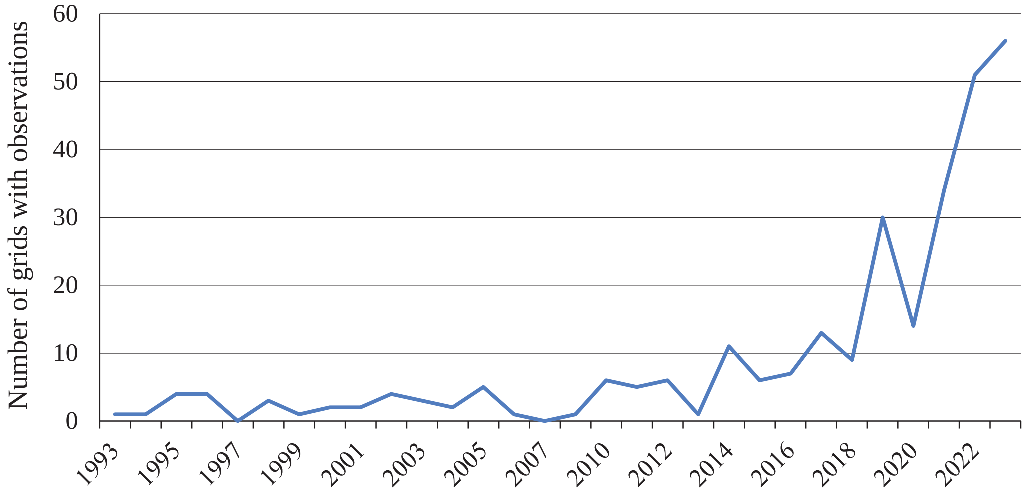

Figure 2. Number of UTM-grids (1x1 km²) with raccoon observations in Flanders.

wear
- Mol2cl: degree of wear on the molars in two classes; 1 = none, 2 = wear present
- Can: closure of the canine root apical foramen; 1 = open, 2 = closing, 3 = closed
- UtW: weight of the uterus in grams
- Pregnancy: P pregnant, BP recently been pregnant (with placental scars), NP not pregnant, NBP not been pregnant
- BacW: weight of penis bone (baculum) in grams
- BacL: maximum length (A2 see Schwery et al. 2011) of penis bone in mm

In addition to the findings from autopsies on collected specimens, the likelihood of reproduction in the wild could also be estimated based on the information associated with other data types, i.e. observations. Specifically, this may involve verified images of living or dead specimens, well-argued sightings of living or dead animals, and catch results.

Results

Raccoon presence

With the oldest validated data dating back to the early 1990s, the INBO carnivore database contains more than 350 observations of raccoons in Flanders validated as 'certain' or 'very probable' for the period up to mid-2024. Figure 2 shows the trend in the number of UTM-grids with raccoon observations over the years. During the first two decades, the numbers fluctuated around a low value, while in the last decade they have increased rapidly and undeniably. This trend is also clear when comparing the corresponding map representations. For the periods 1993-2003 (Figure 3) and 2004-2014 (Figure 4), the coloured grid cells are sparsely distributed and, with a few exceptions, do not overlap. During these periods, no high density of sightings was recorded anywhere. For the period 2015-2024 (Figure 5), the picture is very different, with multiple clustered grid cells and frequently a higher density of observations.

Reproduction

An overview of a series of necropsy results from the collected raccoons with a view to determining possible reproduction in the wild is given in Table 1. As mentioned above the focus is on the reproductive status of the

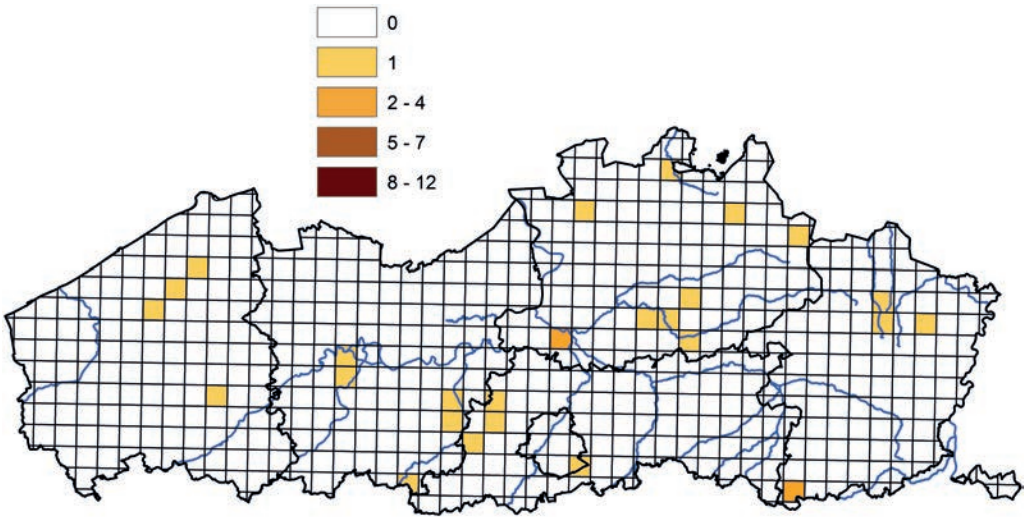

Figure 3. Validated data ('very probable' and 'certain') on raccoon presence in Flanders for the period 1993-2003 based on the 5x5 km² UTM grid.

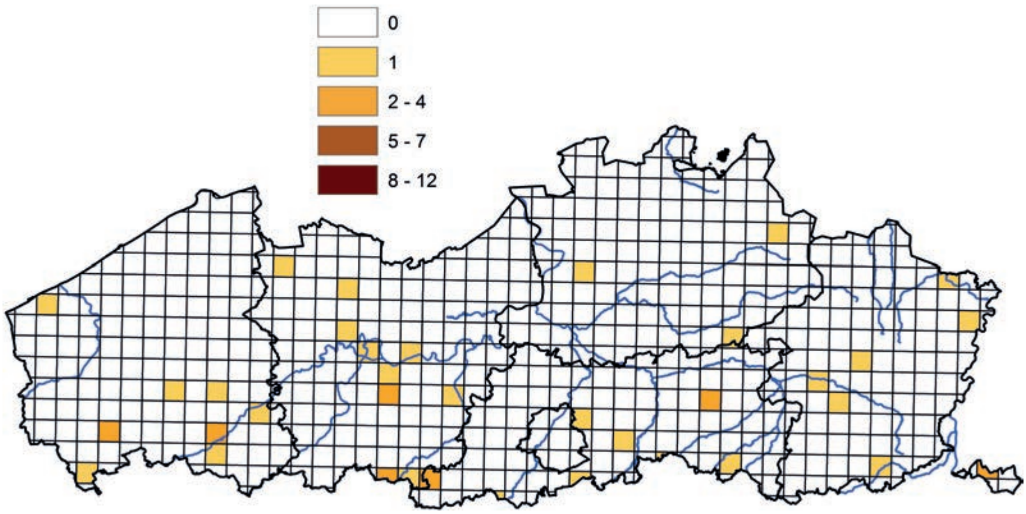

Figure 4. Validated data ('very probable' and 'certain') on raccoon presence in Flanders for the period 2004-2014 based on the 5x5 km² UTM grid.

females and on an approximate age determination with a distinction between juvenile and older (subadult and adult) animals. The values marked in yellow in the table are characteristic of juvenile and subadult animals, i.e. with an age of up to approximately six months and between approximately six months and approximately one year, respectively. The cells marked in green indicate females that are pregnant or have reproduced in the recent past.

Figure 6 shows the locations in Flanders where reproduction in the wild has been observed or can at least be strongly suspected. In fact, reproduction has now been demonstrated in every province; in the Province of West-Flanders there is one location, in East-Flanders there are four locations, in Antwerp

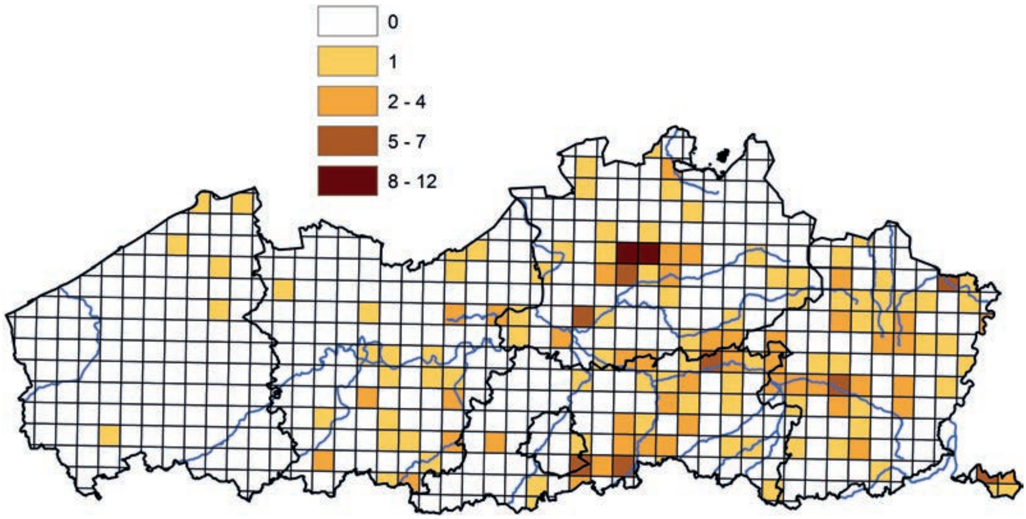

Figure 5. Validated data ('very probable' and 'certain') on raccoon presence in Flanders for the period 2015-mid 2024 based on the 5x5 km² UTM grid.

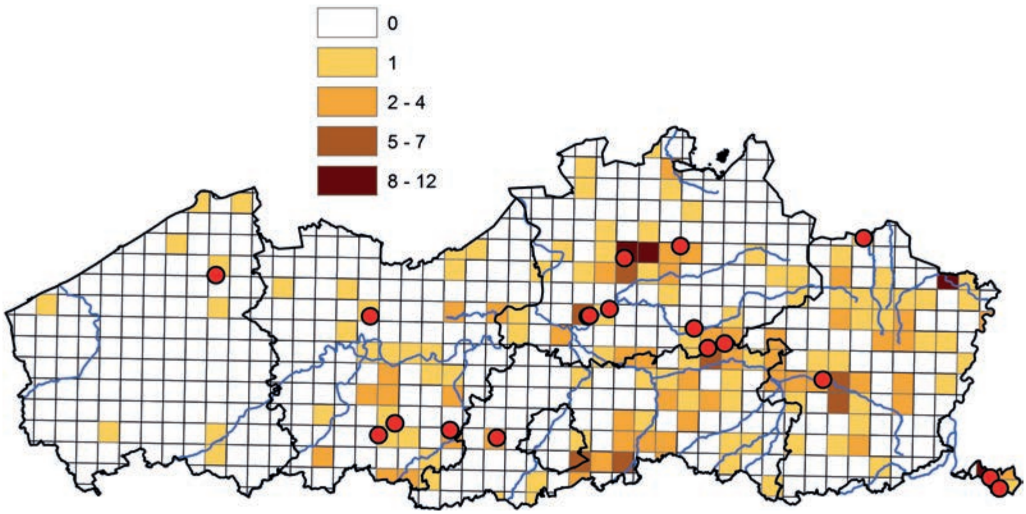

Figure 6. Locations of (very probable) raccoon reproduction (red dots) referenced to all validated data ('very probable' and 'certain') of raccoon presence in Flanders for the period 2010-2024 based on the 5x5 km² UTM grid.

there are seven locations, in Flemish Brabant there is one location, and in Limburg there are four locations. In addition, reproduction was observed in some locations for more than one year.

West-Flanders

Around 20 June 2019, a mother animal with three young was spotted in a garden in Assebroek (Bruges) on a summer evening. After that, no further sightings were reported for that region.

East-Flanders

On 19 February 2017, i.e. in the middle of the mating season, an adult male was collected as a roadkill victim in Grotenberge (Zottegem). A few days earlier, presumably the same spec-

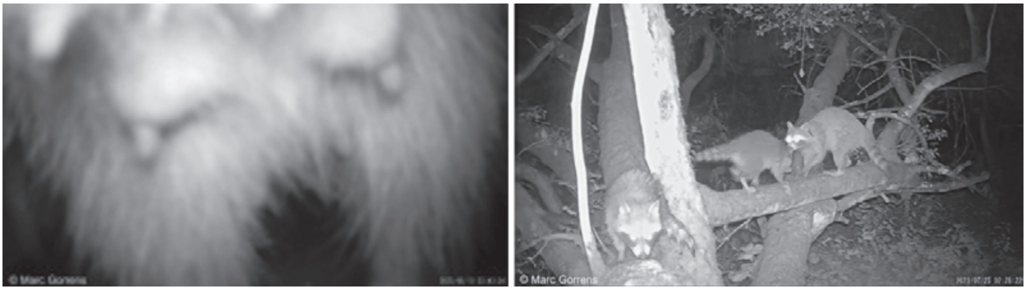

Figure 7. Lactating female raccoon (left) and juveniles (right) in Oelegem (Ranst), respectively on 19/6 and 25/7/2023. *Photos: Marc Gorrens.*

imen was photographed in a garden about five kilometres away. The distance between the two sightings is too great to consider the animal locally resident, but it is quite conceivable that a rutting male (as a 'floating male') could cover such a distance in a still sparse population (Van Den Berge 2017). The animal had a large bald patch on its rump, which is a typical rubbing mark for animals in mating mode.

In late spring 2018, a (obviously) young animal was seen as a roadkill victim on the busy N28 motorway near Ninove. At the beginning of July of the same year, a second and third roadkill victim were found on approximately the same stretch of road.

In Oostakker (Ghent), a recently lactating female was found as a roadkill victim near the harbour area on 10 August 2019. Given that mother animals are normally accompanied by their already fairly large young in the summer, it is unlikely that this was a recent 'stowaway' from a ship.

Antwerp

On 23 September 2015 a young animal was collected (Table 1) as a roadkill victim in Wechelderzande (Lille).

In June and early July 2019, a total of five young animals were captured in Duffel after they had been widely reported in the local press, while a few months earlier adult animals had also been recorded (two on camera, one poisoned).

In Herselt, an early-pregnant female was collected as a roadkill victim on 18 February 2020, and on 20 October 2021 a young male was collected in Westerlo (Table 1). On 9 October 2022, a young specimen was captured in Lier.

In Oelegem (Ranst), a nursing female and later (at least) three young were recorded in the summer of 2023 in an extensive series of camera trap recordings (Figure 7).

Flemish Brabant

In the first half of 2015, an entire family of raccoons was captured and killed in Schepdaal (Dilbeek) (Gouwy et al. 2015, 2016). The family consisted of a subadult male, two adult females, one of which was lactating (Figure 8) and one that must have had cubs in the previous year, and two cubs (Table 1).

On 29 July 2015, a first-year cub was found dead in Hoeilaart. The animal was noticeably thin and had probably been orphaned for some time (Table 1; Gouwy et al. 2015).

A subadult specimen was captured in Averbode (Scherpenheuvel-Zichem) on 3 January 2020, and on 18 December 2022 one was collected as a roadkill victim (Table 1) in Schaffen (Diest). Not far from there, a juvenile was photographed on 19 June 2023.

Limburg

In Hasselt, on 18 January 2018, a female was collected as a roadkill victim that apparently had young in the previous year; on 5 October 2023, a subadult male was collected as a roadkill victim in the immediate vicinity (Table 1).

In Oudsbergen, a young male was collected

Figure 8. Lactating female raccoon, killed in Schepdaal (Dilbeek) in June 2015. *Photo: INBO*.

as a roadkill victim on 22 October 2020 (Table 1), and on 2 August 2022, a mother and two young were observed in Neerpelt.

In Voeren, at two locations, a nest with young was captured in 2021, 2022 and 2023.

Discussion

Population history

Based on the earliest documented introductions, i.c. in Germany (Lutz 1984), raccoons have been present in Western Europe as a wild species for almost a century. Various releases and escapes led to permanent and successful establishment in several regions, initially mainly in Germany and France (see, among others, Salgado 2018). Long-term attempts in Germany to eradicate the species through intensive hunting failed, and the species has remained permanently present in a number of regions for some time (Lammertsma et al. 2008, van der Grift et al. 2016), while also giving rise to a continuous flow of dispersing animals to new regions.

The first documented mention of a free-roaming raccoon in Belgium dates back to 1986 (Libois 1987) and concerned a roadkill found in Amel (Sankt-Vith, Province of Liège). Its appearance in Belgium, close to the German-Dutch border, was not interpreted as a surprise, quite the contrary ("*La présence en Belgique du raton laveur était, à vrai dire, attendue depuis belle lurette*", and "*il est même surprenant que ce carnivore n'ait pas franchi nos frontières avant 1986*"), given that the species was already considered to be regularly occurring in the German and especially Dutch (South Limburg) border areas at that time.

Nevertheless, the question arises, especially in relation to current knowledge about the time frame of further population development and its genetic framework (see below), whether this was actually a 'wild' (dispersing) specimen or just an animal from captivity. Apart from a series of skull measurements (apparently an adult), no supporting information is provided, and neither the sex nor the reproductive status is mentioned. Incidentally, the Royal Saint Hubert Club of Belgium (Koninklijke Sint-Hubertusclub van België 1971) reported a raccoon caught by a gamekeeper in Sart-Eustache (Province of Namur), which also cites a communication with the Royal Belgian Institute of Natural Sciences about another isolated case of a killed animal in the vicinity of Spa (Province of Liège) around 1964. At the time, it was decided that such cases must involve escaped animals, as no pair or family of wild raccoons had ever been observed in Belgium.

For Flanders, the first records relate to the north of Limburg, dating back to the first half of the 1980s (Holsbeek et al. 1986): a specimen shot in 1982 in Molenbeersel-Bree, and a sighting in 1985 in the military domain of Leopoldsburg. However, the species was not (yet) considered to be permanently present at that time, with the individual cases being regarded as possible artefacts (relocated animals).

For the Netherlands, Hoekstra (1983) concludes that, from 1965 onwards, there was also an influx of dispersing raccoons from Germany. It was assumed that the earlier records (dating back to the beginning of the last century) must all have originated from zoos, travelling circuses, fur farms, private individuals, as well as Allied army units that carried raccoons as mascots.

In France, apart from scattered reports from across the country (and therefore most likely all animals originating from captivity), a strong population centre developed in the northern department of Aisne from 1966 onwards, based on specimens that had escaped from an American military base (Léger 1999). For the border region of Nord-Pas-de-Calais, an isolated sighting (capture) in 1992 led to the long-awaited conclusion that "*Le raton laveur est arrive*" (B. 1992). However, this was in all likelihood an escaped animal and, years later, there is still no evidence of a local occurrence of the species (cf. Léger 1999).

In the Flemish Mammal Atlas, Van Den Berge & De Pauw (2003) consider the raccoon to be mainly first-generation escaped animals, with no evidence of local reproduction. The same analysis remains valid in subsequent years, although at the same time it could not be ruled out that there may also have been a few cases of distant dispersers from neighbouring countries (Van Den Berge 2008, Van Den Berge & Gouwy 2009). Somewhat remarkable is an observation in Gingelom (Province of Limburg), where in February 2003, during an (illegal) fox hunt, two raccoons were found together in a burrow, possibly indicating mating.

In a number of cases, the information accompanying a specific report – sometimes an extensive story, often accompanied by press articles – indicates that there is evidence or at least strong indications that these are recent, first-generation escaped animals. Sometimes these indications are obvious, such as when the animals are microchipped or tame, but behaviour alone does not always provide a definitive answer. An interesting report in this regard concerns the recapture of an (individually recognisable) specimen after it had, apparently, lived in the wild for five months without any problems, travelling a distance of at least approximately 35 km and becoming relatively shy in the meantime (De Meulemeester 1995, Van Den Berge & De Pauw 2003). In principle, such cases can therefore give rise to a series of sightings spread over a relatively long period (months). If, every now and then, a specimen escapes 'quietly' from a nearby source, such as an animal dealer and his local clientele, over a period of several years, this can quickly give rise to the suspicion that wild specimens have settled locally. Such a combination of events was probably – given the subsequent discontinuity of data – at the root of a series of observations in the Nete Valley as described by De Smet & Vandewalle (1995).

As mentioned above, the overall appearance of map representations for the periods 1993-2003 (Figure 3) and 2004-2014 (Figure 4) shows a fairly similar picture of scattered, irregular presence. Furthermore, with a few exceptions, the coloured map squares are all different in both time periods and no high density of sightings was recorded anywhere. This suggests that there is little or no connection between the data from these periods and that this data probably refers to random observations, such as escaped and wandering animals.

In contrast, the picture for the period 2015-mid 2024 (Figure 5) is clearly different with multiple clustered grid cells and a higher observation density, confirming the trend previously identified by D'Hondt et al. (2023) for a shorter period of time. So, it's obvious that raccoons became widespread in Flanders during the last decade, as evidenced also by a recent increase in (collected) roadkill (see also Table 1) and camera recordings. The recent upward trend in Flanders can be logically explained in the context of developments in neighbouring regions. On the Dutch side, Akkermans & Mulder (2016) – contrary to Hoekstra's (1983) prediction – stated that the raccoons present at the time did not yet form a self-sustaining population, but that the front of the German source population was gradually approaching the Dutch border. A few years later, La Haye et al. (2021) noted that this front was still some distance away from the border in (Dutch) Limburg, but that a local population could now develop from captivity. In France, and more specifically in the north-eastern departments, a spectacular expansion was taking place (Léger & Ruette 2014, Larroque et al. 2023). Subsequently, large parts of Wallonia were seamlessly colonised within a few years (Schockert 2017).

Reproduction in the wild

As mentioned earlier, determining reproduction in the wild is a classic criterion for distinguishing between the random occurrence of individual animals (stray animals, escaped specimens, etc.) and the possible establishment of a species in a (nascent) population.

In principle, some reservations should always be made regarding the interpretation of the biological characteristics that have been established. For example, the discovery of a nursing female as a roadkill victim does not necessarily mean that the animal's previous reproductive cycle also took place in the wild. It is possible, for example, that a pregnant female escaped from captivity and gave birth to her young in the wild some time later. It is also possible that a mother animal could escape with some of her young, or that one or more young could escape separately. Therefore, local circumstances should also be taken into account as far as possible, such as the proximity of a known location where exotic animals are or were recently kept, or when it appears that (live-captured) animals are exhibiting unnatural tame behaviour.

For the cases examined here, it was not always possible to reach a clear-cut conclusion regarding the certainty of reproduction in the wild. However, through a critical evaluation of the circumstances of each of these cases, it was possible to conclude for the vast majority of them that this was (in all likelihood) indeed the case. Almost all of them were 'surrounded' by numerous other sightings in the more or less immediate vicinity, either in the preceding years or later. Moreover, with the exception of one possible sighting in 2003, all cases date from the last decade. Conversely, the mere observation of raccoons in a particular area for several consecutive years, and certainly when there are occasional indications of reproduction, logically argues for an (emerging) established population core. So, for rapidly expanding species, necropsy findings or other 'hard evidence' can even quickly become outdated compared to field reality.

Trend reversal

Although invasive raccoon expansion has been feared and warned about for several decades in Western Europe, it has only become an issue relatively recently. Since the mid-

1990s, a clear dynamic has emerged in the former strongholds, with the species (only then) beginning the steep part of the classic S-shaped population growth curve and numbers subsequently increasing exponentially (Salgado 2018). Accurate information on European distribution is available via the reporting mechanism of the EU Regulation. Raccoon is the most widespread species in the EU Regulation based on the number of 10x10 kilometre grids in Europe (>6500 grids, Tsiamis et al. 2017). The expansion front has now reached Flanders.

It is remarkable how the raccoon population trend fits in with the analogous trend that has occurred among various native carnivores in recent decades (cf. Van Den Berge & Gouwy 2021). Whereas native predator species are in a recovery phase after a long period in which the respective species had become locally or quasi extinct, an alien species such as the raccoon is, of course, experiencing a new emergence. The causes of these apparently prolonged slowdowns, despite their widespread and latent presence prior to the respective recent population breakthroughs, are not precisely known. A general change in environmental and nature policy, including an international ban on the use of non-humane (but efficient) hunting and control methods such as certain types of traps and heavy poisons, has at least facilitated this general trend reversal – if not caused it.

Stricter European environmental standards, including the ban on the production and use of PCBs since 1985, have probably also played an important role here. These substances are particularly toxic in terms of reproductive disruption and genetic abnormalities (He et al. 2021). For several decades, the bioavailability of these substances and other fat-soluble toxic contaminants (dieldrin, heavy metals, etc.) in the Western European food web has been declining (see, for example, de Boer et al. 2010). For semi-aquatic predators such as otters and American minks, such pollutants have long been considered genuinely harmful (see, for example, Jensen et al. 1977, MacDonald & Mason 1994, Van Den Berge et al. 2019). These substances may also have been the reason why no wild population of American mink was able to develop in Flanders (and the Netherlands) (Van Den Berge & De Pauw 2003, Dekker 2012). The raccoon, whose diet consists largely of aquatic organisms, is also one of the species considered to be particularly sensitive to PCB concentrations in the environment (U.S. Environmental Protection Agency n.d.).

In addition, more generic aspects of habitat improvement may also have played a facilitating role in population development. For example, the trend towards more nature-oriented forest management, including attention to ageing trees and more dead wood (Pro Silva Europe s.d., Forest Europe 2020), will have led to an increase in good shelter and nesting sites. The recovery of aquatic biodiversity (Haase et al. 2023), including an increase in macroinvertebrates, will have led to more food. The widespread occurrence of alien species, in this case crayfish, often in high densities, is also linked to the emergence of the raccoon (Salgado 2018).

Genetic context

Ongoing genetic research – for which INBO also provided samples, see Frantz et al. (2013), Fischer et al. (2015), Maillard et al. (2020) and Larroque et al. (2023) – has now shown that the origin of local animals is often more complex than previously assumed. In addition, for the wider Western European region, there are indications or findings of relatively recent (i.e. in the last few decades) multiple new inputs of raccoons of different genetic origins from captivity. Noteworthy in this context is also the finding by Maas et al. (2021) that the Dutch raccoons in South-Limburg could be characterised as a largely separate genetic group (at least until a few years ago), originating locally from captivity.

Currently, the respective, originally isolated Western European areas are becoming interconnected through both steady expansion at the periphery and, in some cases, long-distance dispersal (up to 200-300 km, Michler 2018, Larroque et al. 2023) of individual animals. Belgium occupies the geographical centre between the original French and German areas and thus forms the contact zone between them. In this context, it can be assumed that the increasing genetic mixing associated with this will also promote the genetic fitness of the species in Western Europe. This, in turn, may reinforce the purely mathematical effect of the exponential growth phase in which the population currently finds itself – or at least limit the risk of a possible (temporary) slowdown in growth due to potential inbreeding effects.

A remarkable, albeit (perhaps very) exceptional occurrence concerns the possible introduction of new and originally indigenous genetic material via animals that are transported as 'stowaways' by ship. In 2014 (Gouwy et al. 2014) and 2021 (D'hondt et al. 2023), for example, a raccoon was found in the ports of Antwerp and Zeebrugge respectively on a ship that had departed from the United States.

Afterthoughts

Double standards?

It may be taken for granted that no one concerned with nature conservation and biodiversity will welcome the arrival and establishment of a new non-native species. However, with regard to non-native carnivores, a warning should be issued about the risk of applying an overly nuanced double standard. Most native carnivore species have also recently made, or are in the process of making, a remarkable comeback, without existing communities (or the prevailing perception thereof) being 'prepared' for it. The return or population recovery in Flanders of species such as the fox and stone marten (just a few decades ago) or the pine marten, badger, otter, and wolf (currently) was or is generally welcomed with great enthusiasm by nature conservationists, as the final element of rich communities. The question then arises as to what the specific problem is with regard to non-native carnivores, and, in other words, what is not an issue with native species.

Classically, a whole range of undesirable characteristics are cited regarding the raccoon, all of which call for drastic intervention. These range from predation on domestic animals and vulnerable wild prey species, the spread of zoonoses, to damage and nuisance in buildings. These 'adverse characteristics' however, all apply equally to native predator species. Regarding native species, we generally advocate a completely different approach, partly due to the special position of predators as keystone species in ecosystems (see, for example, Agentschap voor Natuur en Bos 2014). Chickens, pigeons, and sheep should be protected from being killed by foxes, martens, or wolves with appropriate fencing. Predation on nests of, for example, black woodpecker (*Dryocopus martius*) and honey buzzard (*Pernis apivorus*) by pine marten is 'part of the game', and there is now a broad consensus that important breeding sites of meadow birds can be protected from excessive predation by terrestrial predators, mainly by fencing (see, among others, White & Hirons 2019, Teunissen et al. 2020). Regarding the risk of transmission of the fox tapeworm (*Echinococcus multilocularis*), one of the most threatening parasitic challenges in the European Union (EFSA & Zencanaro 2019), from foxes (or wolves) to humans, we limit ourselves to focusing on basic hygiene and the sometimes very low local prevalence of the parasite. Regarding damage and nuisance in buildings, we expect (potential) victims to take the necessary preventive measures to prevent stone martens from entering or accessing buildings. It should be noted that once a marten can no longer get in somewhere, this will certainly not be possible for the larger raccoon either.

Essential in considering potential problems and nuisance caused by predators is the message that the correct handling of them is not dictated simply because, in the case of native carnivores, they are protected species. After all, a species' legal status is the subject of a societal choice, and therefore nothing more than the result of a human decision. It is not this legal status, and therefore not the prohibition on capture or killing, that is the cause of potential problems or inconveniences, but (literally) the nature of the creature. In other words, the recommended solutions are not forced alternatives to control (killing) because they concern protected species: they simply offer the most realistic, sustainable, and effective solution. Anyone who kills a fox or wolf to protect their chickens or sheep cannot avoid the risk that sooner or later a new fox or wolf will appear, which will in turn attack the unprotected chickens or sheep. Anyone who captures a stone marten after the roof insulation of their home has been destroyed can very likely expect a recurrence of the incident if martens continue to have free access. Controlling foxes to reduce the risk of fox tapeworm can be counterproductive (see, for example, Comte et al. 2017).

Is culling recommended?

Previous analyses (see Van Den Berge & Gouwy 2021 for an overview) have explained that for year-round territorial predator species, with their generally naturally capped low densities, indiscriminate culling of individuals would only be meaningful if control leads to (near) extermination of the species across a very large area. However, with currently permitted – humane and selective – means and methods, this often proves impossible in practice once the species has established a population. While eradicating native predator species is no longer a viable objective, the question remains whether it is even possible for non-native species, using (solely) humane and selective means (cf. Smith et al. 2022). Documented significant impact on biodiversity should be the basis for such a decision (Davis et al. 2011). Moreover, if the proposed objectives prove difficult or impossible to achieve, for example, due to compensatory population demography, the ethical aspects of the applied management become all the more pressing. The systematic killing of (highly developed) animals can then hardly be justified solely on the grounds that a non-native species is involved.

Despite successful examples of near-eradication of nascent raccoon populations (e.g. Mazzamuto et al. 2020), it is unrealistic, in this phase of growth and within the given geographical context, to prevent further expansion and establishment of raccoons in Flanders. Given the geographic position relative to immediately neighbouring, already occupied areas in northern France and Wallonia, as well as in Germany, continued recolonization can be expected. In this context, D'hondt et al. (2022) already point out that comprehensive management on a Flemish scale is not evident due to the problematically low perspective for action, which is based on, among other things, the high risk of recolonization from neighbouring countries and regions (Adriaens et al. 2019).

A critical yet comprehensively argued analysis of the current standard raccoon control policy (in the Netherlands) is provided by Mulder (2008, s.d.). In it, virtually all the prevailing motives for raccoon control are refuted or, in their broader context, significantly nuanced (including the public health risk associated with the raccoon roundworm *Baylisascaris procyonis*). At the same time, explicit reference is made to the previous, extensive, but ultimately unsuccessful control efforts in Germany, from where the raccoon range is expanding unstoppably towards the Netherlands. The final message is that it will be a matter of learning to live with the raccoon as a new species, whether one wants to or not.

Staying focused

An interesting point of attention within this issue remains the raccoon's further population dynamics, coupled with its complex social organization. Somewhat unlike our native medium-sized predator species, the raccoon sometimes appears to live in loose groups with conspecifics: clusters of females and 'coalitions' of males (Michler 2018). In rural areas, densities can potentially be reached that are two or three times higher than those of our native species, although, so far, this has only been observed in the native range. In urban areas, numbers sometimes reach (very) high densities (Fischer et al. 2016). While this could be an argument against allowing it to get to this point, this inherent population plasticity could also fuel the argument that comprehensive (expensive) control measures (only using humane means and methods) will have little effect in practice.

In certain cases, temporary and localized management may be appropriate, with prevention of nuisance and impact being the primary management drivers, and control measures implemented only as a last resort, aimed at safeguarding specific natural values. This can potentially save time locally, for example, by reducing the vulnerability of a potentially endangered prey species through appropriate habitat restoration. Habitat restoration generally requires sufficient time, assuming that it is effectively implemented in the meantime. The latter generally remains the foundation for nature restoration and therefore requires the necessary commitment and budgets.

Acknowledgements: The operating 'Marten Network' could rely on support from the Flemish Agency for Nature & Forests (ANB) and the Royal Belgian Institute of Natural Sciences (KBIN/IRSNB). We gratefully acknowledge the volunteers of the Marten Network for their help in collecting the dead specimens and the Flemish shelters for birds and wild animals for sharing their data on raccoon captures. Many thanks to an anonymous reviewer for useful suggestions that improved the text, and to Marc Gorrens for the use of his photos.

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Samenvatting

De opkomst van de wasbeer (*Procyon lotor*) in Vlaanderen: kroniek van een aangekondigde evolutie?

Het voorkomen van de wasbeer als uitheemse soort in West-Europa wordt al enkele decennia met bijzondere aandacht gevolgd. Via het Marternetwerk van het Instituut voor Natuur- en Bosonderzoek (INBO) wordt in Vlaanderen sinds 1998 werk gemaakt van een systematische opvolging van wasbeeraanwezigheid in het wild. Daarbij is gebleken dat het bij de waarnemingen uit de jaren 1980, 1990 en beginjaren 2000 zo goed als allemaal om eerste-generatie ontsnapte of door mensen getransporteerde dieren moet zijn gegaan. Nergens kon spontane vestiging, met voortplanting in het wild, worden aangetoond. Sinds zowat een decennium is daarin overtuigend verandering gekomen. Op basis van de integratie van alle waarnemingen met bijhorende informatie, waaronder ook de bevindingen omtrent de voortplantingstoestand van ingezamelde verkeersslachtoffers, blijkt dat de populatieontwikkeling zich verspreid over geheel Vlaanderen reeds in een gevorderd stadium bevindt. Deze nieuwe toestand is het logische uitvloeisel van een opgemerkte dynamiek, die sinds het midden van de jaren 1990 in het vroegere Duitse en later ook Noord-Franse bolwerk op gang is gekomen. Actueel is de Waalse regio bezuiden de Samber-Maaslijn naadloos vanuit de Franse en Duitse grensregio's als het ware 'dichtgevloeid' met wasberen, en spreidt deze kolonisatiegolf zich geleidelijk aan verder uit over Vlaanderen en Henegouwen. Deze trendbreuk is opmerkelijk maar niet uitzonderlijk in West-Europa. Zij spoort samen met een analoog gebeuren bij meerdere (middel)grote (zoog)diersoorten als een gecombineerd effect van afgenomen actieve doding (jacht, bestrijding) en toegenomen habitatkwaliteit, waaronder o.a. afname van zwaar-toxische contaminanten in de voedselketen en gerichte natuurherstelmaatregelen. Als niet-inheemse soort is de wasbeer in principe ongewenst in de natuur. Specifiek voor de soort zelf worden daarbij verschillende redenen aangehaald: het betreft een predator die mogelijk een bedreigende invloed uitoefent op de biodiversiteit, regelmatig pluimvee doodt en gebouwen beschadigt, en drager kan zijn van een voor de mens niet-ongevaarlijke parasitaire worm. Over de werkelijke draagwijdte en impact van deze negatieve eigenschappen bestaat in wetenschappelijke kringen discussie. Uit langjarige beheerervaring en dito -onderzoek in Duitsland is gebleken dat wasberen zich, eens stevig gevestigd, heel moeilijk efficiënt laten bestrijden en terugdringen. Het is dan ook de vraag of, en zo nodig hoe, wasberen (nog langer of opnieuw) systematisch dienen bestreden te worden in West-Europa. Los daarvan kan ingrijpen in specifieke situaties wenselijk zijn om, lokaal en tijdelijk, een acuut natuurbehoudsprobleem te helpen milderen of te vermijden. Dit artikel is gebaseerd op het eerder verschenen INBO-rapport van Van Den Berge et al. (2024).

Received: 27 October 2025
Accepted: 6 November 2025

Habitat suitability for the reintroduction of the European mink (*Mustela lutreola*) in the Netherlands

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Abstract: As one of the most critically endangered mammals in Europe, the European mink (*Mustela lutreola*) is in need of urgent conservation actions to ensure its survival. With the disappearance of the American mink (*Neogale vison*) in the Netherlands, one of the European mink's primary threats has been eliminated here, creating an opportunity for its reintroduction. The aim of this study was to identify the potentially most suitable areas for a reintroduction in the Netherlands. To this end, a rule-based habitat suitability model was created based on land use, natural waterways, road density and human presence. The Netherlands seems to have a considerable number of suitable areas for reintroduction, although their favourability varies depending on factors such as fragmentation and muskrat (*Ondatra zibethicus*) control efforts. Wetland areas in the north of the Netherlands were generally identified as the most suitable for a reintroduction, with the Weerribben-Wieden and wetlands at the border of the provinces of Groningen and Drenthe as standout areas. Lowland peat areas in the west with high densities of American crayfish (*Procambarus clarkii* and *Procambarus acutus*) are also good options, although fragmentation and the high intensity of muskrat trapping present notable challenges here. Further investigation into the potential release sites is recommended to examine habitat characteristics in greater detail, assess connectivity and get a grip on the local political and societal dynamics that will play an important role in the success of a reintroduction.

Keywords: European mink, *Mustela lutreola*, reintroduction, habitat suitability analysis, the Netherlands.

Introduction

Biodiversity worldwide is declining due to a range of factors, one of which is the spread of invasive species (Mollot et al. 2017). While some mammals in Europe are recovering due to legal protection and improved habitat conditions, others continue to decline due to the impact of invasive species. One such example is the European mink (*Mustela lutreola*), which is considered one of Europe's most endangered mammals and which is listed as Critically Endangered on the IUCN Red List (Põdra et al. 2025). It is protected as an Annex II and Annex IV species in the EU Habitats Directive. The main threat to the species is the invasive American mink (*Neogale vison*), which was brought to Europe for the fur trade, where it has driven the European mink to near extinction. As a larger and more adaptable species, the American mink is able to outcompete its European counterpart, both through direct competition and interference competition (Maran 2007, Santulli et al. 2014, Põdra et al. 2025). Moreover, it is a known carrier of Aleutian disease virus (ADV), to which the European mink is particularly vulnerable (Maran 2007). Importantly, the disappearance of the European mink in many areas cannot be solely attributed to the presence of the American mink. However, by occupying its niche, the American mink has prevented a resurgence of the European mink despite improved habitat conditions and legal protection (Maran & Henttonen 1995).

In many European countries the American mink has become widespread and has taken over the niche of the European mink. As a result, the European mink now occupies only a fraction of its original range, with confirmed current occurrence only in southwestern France, northern Spain, the Danube delta in Romania and Ukraine, and the Dniester delta and the Carpathians in Ukraine (Harrington & Maran 2024, Mitchell-Jones et al. in press). In addition, there is an introduced population on the island of Hiiumaa in Estonia and there are some isolated populations in Russia (Harrington & Maran 2024). All populations are highly isolated and increasingly threatened by the encroachment of the American mink (Põdra & Gómez 2018).

Many countries are struggling to find effective solutions to control the American mink. In the Netherlands however, the American mink revealed an interesting trend in 2021. After the accelerated closing of American mink farms in 2020 due to the spread of COVID-19, the population of feral American minks showed a rapid decline. By 2021, the American mink had almost been completely eradicated from the Netherlands (La Haye & Janssen 2025). Small founder populations, combined with intensive control using muskrat (*Ondatra zibethicus*) traps are likely the reasons that the American mink has never been able to build up a substantial population in the Netherlands. The closing of the mink farms meant that there was no more influx of new individuals, resulting in extinction on a national scale.

Given the critically endangered status of the European mink, it is expected that the species will not persist without serious conservation actions (Maran 2007, Põdra et al. 2025). So far there have been reintroduction programmes in Estonia and Germany. In Estonia, the island of Hiiumaa was cleared from American mink, followed by European mink releases between 2000 and 2016 (Maran et al. 2009). A stable population size has since been observed. In Germany, reintroductions were done in Saarland (Peters et al. 2009) and near Steinhuder lake (Lange et al. 2022), but little is known about the state of these populations.

The rapid decline of the American mink in the Netherlands (La Haye 2025) sparked an idea: with one of the main threats to the European mink almost having vanished, could this open the door for a reintroduction of the European mink? This led to a feasibility study carried out by Lange et al. (2022), in which they studied the possibility of reintroducing the European mink in the Netherlands, based on the IUCN guidelines for reintroductions. They concluded that most threats to the European mink in the Netherlands have been mitigated, but scrupulous control of American mink remains essential. They argued that the European mink is unlikely to pose a large risk to native Dutch species such as meadow birds, as predation of European mink on medium-sized birds is limited. Although some competition with other mustelids may occur, the European mink's diet differs substantially from that of species like the stoat (*Mustela erminea*) and the western polecat (*Mustela putorius*). Moreover, the species coexists with these relatives throughout much of its current range, suggesting that interspecific competition would be minimal. Lange et al. also argued that the Netherlands, with its many wetlands, supposedly offers very suitable habitat for the species, as it is a riparian mammal. The Biesbosch and Weerribben-Wieden were proposed as potential candidate areas. However, the importance of a habitat suitability analysis was emphasized in order to identify the most promising areas for a reintroduction. This study aims to address that need.

The Netherlands is currently among the

lowest ranking countries of the EU member states regarding the state of its nature and it is not on track to meet its biodiversity targets (Nationaal Dashboard Biodiversiteit 2025). As a predator high in the trophic system, the European mink would therefore be a valuable addition to the Dutch river delta ecosystem. The species may also provide an important ecosystem service, as it has been found to exhibit signs of crayfish preference (Haage et al. 2017). It could therefore potentially aid in the control of the invasive red swamp crayfish (*Procambarus clarkii*) and white river crayfish (*Procambarus acutus*) in the Netherlands.

Youngman (1982) has critically reviewed the information on the distribution of the European mink. His conclusion for the Netherlands is that there is only one proven record: a skull from about 2300 BC found in a hunting camp at Vlaardingen (Van Bree 1961). This is more than four millennia before present. However, based on oral reports from fishermen, Van Bree (1976) suggests that the European mink has disappeared from the Netherlands towards the end of the 19th century. A mounted specimen in the collection of the Übersee-Museum in Bremen, shot in the area of Blockland near Bremen in 1884, further supports the species' presence in the wider region during the 19th century. Blockland is about 110 kilometres from the Dutch-German border. Confusion with polecats may be a reason for the scarcity of historical records of European minks in the Netherlands, as this reportedly also occurred in eastern Europe (Dr. Tiit Maran, personal communication).

Given its critical conservation status, the European mink presents an excellent opportunity for the Netherlands to fulfil a moral obligation and contribute to the recovery of a globally threatened species. Species reintroductions however require high investments, which means that careful planning is crucial. An important first step is identifying the most suitable areas for reintroduction. Predictive models can be used for this, thereby optimizing reintroduction efforts, increasing the likelihood of long-term success (Hunter-Ayad et al. 2020). In this study, we conducted a habitat suitability analysis for the European mink in the Netherlands, providing a comprehensive overview of potentially suitable areas for its reintroduction. This is the first habitat suitability analysis for the European mink in the Netherlands, and no comparable habitat suitability studies from other countries have been published for this species to date.

Methods

Model choice

Habitat suitability modelling can be defined as the application of the ecological niche concept by predicting the likelihood of a species' occurrence based on environmental variables (Hirzel & Le Lay 2008). Recent years have seen the rise of habitat modelling using presence data by relating current species occurrence to environmental variables (Phillips et al. 2004). This is however not possible for the European mink in the Netherlands due to its current absence and therefore a lack of presence data. Extrapolating from the current distribution of the European mink is also not suitable, because its distribution is not only determined by suitability of habitat, but also – in some areas perhaps even more so - by the absence of American mink (Maran & Henttonen 1995). In addition, the Dutch landscape has been modified considerably, which means that a release area may constitute a relatively new environment. Therefore, correlative models may underestimate habitat suitability if the habitat is good but falls outside the range of the data used to train the model (Hunter-Ayad et al. 2020). Past American mink presence in the Netherlands could be used as a proxy for European mink, but the distribution of American mink in the Netherlands correlated strongly with the locations of mink farms, rather than the most favourable habitat (Dekker & Hofmeester 2014).

To the best of our knowledge, there are no habitat suitability analyses for European mink that can serve as an example for this study. For the reintroduction in Estonia, the predicted suitability of the island of Hiiumaa was estimated based on surveys on prey and shelter availability combined with expert judgement (Dr. Tiit Maran, personal communication). Furthermore, the presence of American mink was an indication that the island would be suitable for European mink after eradication of American mink.

For the habitat suitability analysis in this study, we used a rule-based model. This is a model based on habitat preferences derived from literature combined with expert judgement. Rule-based suitability modelling has been applied in marine and aquatic suitability modelling, but also increasingly for terrestrial species (Gwynn & Symeonakis 2022).

Habitat preferences

The most important habitat preferences for European mink were identified based on literature and expert judgment, with input from Dr. Tiit Maran (European mink expert) and Dick Klees (carnivore expert, Studio Wolverine). These habitat preferences are:

Land use. European minks are restricted to natural habitats, which we define as areas with minimal human disturbance and a high presence of natural vegetation. They particularly favour wetland habitats with small woody vegetation, such as bramble bushes and reeds, which provide cover and are commonly used for nesting (Zabala et al. 2003, 2006, Palomares et al. 2017, Lange et al. 2022). In some studies, they have been found to avoid dense forests (Zabala & Zuberogoitia 2003, Fournier et al. 2008).

Natural waterways. As a semi-aquatic animal, the European mink is mainly active in the riparian zone. They settle their home range along waterways (Fournier et al. 2008, Palomares et al. 2017) and are mostly found less than 150 metres from a watercourse (Danilov & Tumanov 1976), but at times they can go up to 500 metres (Dr. Tiit Maran, personal communication). Palomares et al. (2017) found that females had small home ranges mainly in lagoons and small tributaries, whereas males also included river sections in their home ranges. The species has been found to be absent from canalised streams, likely due to a lack of shelter and prey sources (Maran & Henttonen 1995, Lodé 2002, Zabala et al. 2006). They are also prone to fragmentation, as their presence has been found to depend on non-fragmented main river stretches and the number of waterways free from barriers (Lode 2002, Zuberogoitia et al. 2013, Goicolea et al. 2022). Mink can reside near fast-flowing or slow-flowing water, provided the water remains unfrozen during winter (Dr. Tiit Maran, personal communication).

Roads. As with many mammal species, roads pose a serious risk to European mink. In Spain, road kills were the most common cause of death in the period 1990-2008 (Palazón et al. 2012). After trapping, the main mortality cause in western France was road kills in the period 1965-1997 (Lodé et al. 2001). In the Netherlands, road mortality is an especially large problem, as is illustrated by the otter (*Lutra lutra*): road mortality was the cause of death in roughly 87% of deceased otters found in the Netherlands (de Groot et al. 2023). European minks usually avoid passing through culverts, just like otters. This means that when moving along waterways they will cross roads over land which leads to road casualties (Dr. Tiit Maran, personal communication).

Human presence. The European mink is sensitive to human disturbance and therefore requires large undisturbed areas. Ortiz-Jiménez et al. (2021) found that European minks hid themselves for longer periods of time when being exposed to anthropic noises, as well as to the odor of dogs. However, due to their nocturnal and crepuscular activity pattern (Garin et al. 2002), the effect of daytime disturbance (e.g. recreation) may be limited.

Table 1. Habitat preferences used in the habitat suitability model.

Habitat preference	Included in model as	Unit	Data type	Year	Sources
Land use	Land use	Categorical	Raster (5x5m)	2023 & 2022	LGN 2023 (https://lgn.nl/bestanden) RIVM Natura2000+NNN (https://www.atlasleefomgeving.nl/groenkaart-van-Nederland)
Natural waterways	Shoreline length	km/km^2	Polylines	2024	PDOK TOP10NL (https://www.pdok.nl/datasets)
Roads	Road density	km/km^2	Polylines	2024	PDOK TOP10NL (https://www.pdok.nl/datasets)
Human presence	Building density	m^2/km^2	Raster (5x5m)	2024	PDOK TOP10NL (https://www.pdok.nl/datasets)

Additional habitat preferences are recognized, but these were not included in the model for the following reasons:

Water quality. This can have a significant effect on especially carnivores due to bioaccumulation of pollutants (Borgå 2013). In the past the level of PCB pollution has been a threat to otters in the Netherlands (Broekhuizen & de Ruiter-Dijkman 1988). However, since the reintroduction of the otter (2002-2008) a viable population has established and the otter is rapidly increasing (de Groot et al. 2023). Apparently water quality is no bottleneck anymore for the otter in the Netherlands. Since European mink is less dependent on fish in its diet than otter, the effect of water quality could be even smaller and therefore it was not included in the model.

Prey availability. European minks have a varied diet consisting of crayfish, fish, voles, mice and more (Lange et al. 2022). Accurate nationwide occurrence data for most of these species is lacking, however. The wetlands of the Dutch river delta are characterized by a high primary productivity and mild winters, which results in high abundances of e.g. mice and crayfish. Therefore, it was assumed that prey availability is not a limiting factor in this model.

Absence of American mink. since the species has almost completely been eradicated in the Netherlands and the last remnant populations are expected to disappear soon, it was not included in the model.

The habitat preferences were included in the model as shown in Table 1. Land use and shoreline length per km^2 together are a proxy for potentially suitable habitat. Road density and building density were included as proxy variables for anthropic disturbance. We opted for building density instead of population density, as this better reflects the effect of recreation by including e.g. harbours and holiday parks.

The Netherlands must adhere to the Berne convention and several key EU regulations and directives aimed at the conservation of wildlife, including the Birds Directive, the Habitats directive and the Invasive Alien Species Regulation. To this end, 162 areas in the Netherlands have been designated as Natura 2000 sites, aimed at conserving the most valuable and threatened species and habitats. We chose not to include Natura 2000 sites as a direct input variable in the habitat suitability model, as their designation is already based on ecological value and could therefore bias the results by inflating suitability scores. Natura 2000 areas were considered afterward to evaluate how well these align with suitable habitats and to identify potential gaps for reintroduction planning.

Data preparation

The data used can be found in Table 1. The

Table 2. Weighted input maps for the habitat suitability model.

Input map	Pixel size	Transformation		Weight
Land use	100x100m	Unique categories:		3
		Land use type	Suitability	
		Buildings	1	
		Agriculture	3	
		Forest	7	
		Dunes	5	
		Heather / drifting sand	5	
		Raised bog	3	
		Other wetland vegetation	9	
		Reed vegetation	10	
		Forest in wetland	10	
		Shrub vegetation in wetland	10	
		Naturally managed agricultural land	9	
		Other grassland	6	
		Other shrub vegetation	8	
Shoreline length	500x500m	Continuous function: MSLarge		2
Road density	500x500m	Continuous function: MSSmall		1
Building density	500x500m	Continuous function: MSSmall		1

data was prepared in QGIS 3.36 Maidenhead. For land use, the LGN2023 map was simplified to only relevant categories using the reclassify tool. Because many Natura 2000 and NNN (Nature Network Netherlands) areas were classified in LGN as agricultural, a Natura 2000 + NNN map was added using the raster calculator. These areas were then classified as naturally managed agricultural land. Water surfaces were excluded from the land use map, since only shorelines are suitable, and these are already included in the shoreline length layer. The land use layer was resampled to a 100 x 100 m raster using the Warp tool. For shoreline length and road density, total length per 500 x 500 m grid cell was calculated using the Sum line lengths tool. Building density per 500 x 500m grid cell was calculated using Zonal statistics.

Suitability modelling

The habitat suitability maps were produced using the Suitability Modeler environment from the Spatial Analyst package in ArcGIS Pro 3.4. This tool transforms input maps into separate suitability maps which are then combined into one final suitability map. The input maps were transformed and weighted according to expert judgment, with input from Dr. Tiit Maran and Dick Klees, as detailed in Table 2. The weights for the input maps and suitability scores for the land use map are continuous, meaning that a land use type with score 9 is considered three times more suitable than a land use type with score 3. For road density and building density the input map was transformed using the MSSmall function (Minimum Slope to Smallest Cell). This function rescales input data based on the mean and standard deviation where smaller values in the input raster have higher preference. This transformation fits well for roads and buildings, as it helps to emphasize areas with low road and building density. For shoreline length the opposite function was used: MSLarge (Mean Slope to Largest Cell). This was done to emphasize areas with a high shoreline length per km^2. A sensitivity analysis was performed by testing the effect of weight variations on the output map.

Identification of suitable areas

Suitable areas were identified using the Locate tool in the Suitability Modeler. Here, the top 5, 10 and 15 contiguous areas were selected based on the highest suitability values for such areas. For the areas, a minimum of 50 km^2 was chosen. In addition, the tool requires an average size for the areas, which was set at 100 km^2. It is difficult to determine the minimum area size based on the requirements of a mink population, as this depends more on the total length of banks of watercourses and prey availability within the area than on the overall size of the area itself (Fournier et al. 2008). In this study, areas of at least 50 km^2 were generally considered sufficient, as these include the most important Natura 2000 sites in the Netherlands. For the shape of the areas, a pentagon was selected with 25% shape/utility trade-off. This setting balances two key objectives: maximizing ecological utility, by allowing the shape to better conform to landscape features such as linear fluvial areas (e.g. brook valleys) while maintaining a roughly circular form. A higher trade-off value (closer to 100%) would result in a more compact, circular shape, while a lower value (closer to 0%) would allow the shape to stretch more freely but potentially lead to highly irregular and narrow forms, which may be less desirable for a reintroduction site.

Results

In Figure 1A, the habitat suitability model for the Netherlands is displayed. The red areas, which are the least suitable, are mostly urban areas (Figure 1D). The yellow parts mostly consist of intensive agriculture. Some agricultural areas in the west show up as light green, as a result of the high shoreline length in these areas (Figure 1C). Natural areas are generally green, with drier areas like the Veluwe being light green, whereas the wetter lowland areas are darker green (most suitable). The more elevated south and east of the Netherlands are generally unsuitable as it is too dry. As such, the 'wet axis' is clearly present in the model, reflecting the wet lowlands. Striking are the green areas in the north of the country. The lowland peat areas in the west of the Netherlands also stand out. Rivers, brook valleys and large lakes are not clearly distinguishable, possibly due to their relatively low shoreline length per square kilometre compared to wetland areas with many brooks and canals.

Scenarios for most suitable areas

Figure 2 shows scenarios for the most suitable areas according to the model. The north of the Netherlands seems to have the most suitable areas in close proximity to each other (Figure 2), suggesting that this could be a good option for a metapopulation. The top five areas are further discussed in Table 3, along with three other areas (labelled A, B and C in Figure 2) that are potentially interesting. Some areas are fully designated as Natura 2000 sites under the Habitats Directive, while others are only partially included or fall solely under the Birds Directive (Natura 2000 Gebieden (LVVN, retrieved on 5-9-2025). Inclusion in the Habitats directive is favourable for a reintroduction, because then specific conservation measures will need to be taken for the European mink, like habitat restoration and reduction of human disturbance.

Discussion

This study has provided a comprehensive overview of the suitability of the Netherlands for a reintroduction of the European mink. The model output serves as a good baseline for

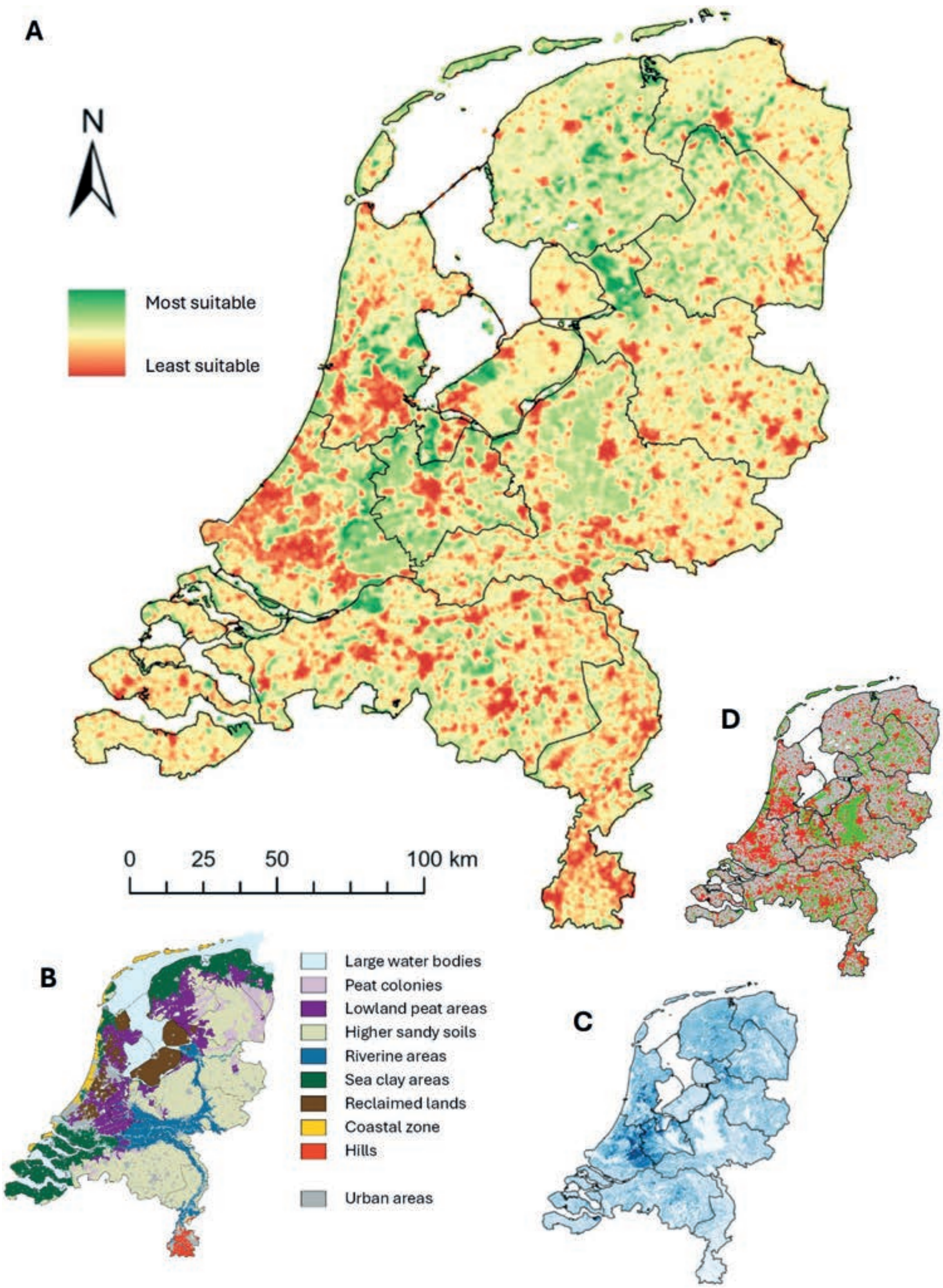

Figure 1. (A) Habitat suitability map for the European mink in the Netherlands. Suitability is based on land use (weight 3), shoreline length per km^2 (weight 2), road density and building density (both weight 1). Red areas indicate the poorest habitat and green areas show the most suitable habitat. (B) Reference map of physical-geographical regions of the Netherlands (source: Statistics Netherlands). (C) Map of shoreline length per km^2, with darker areas having a higher shoreline length per km^2. (D) Simplified land use map of the Netherlands, with red being urban areas, green natural areas and grey agricultural areas.

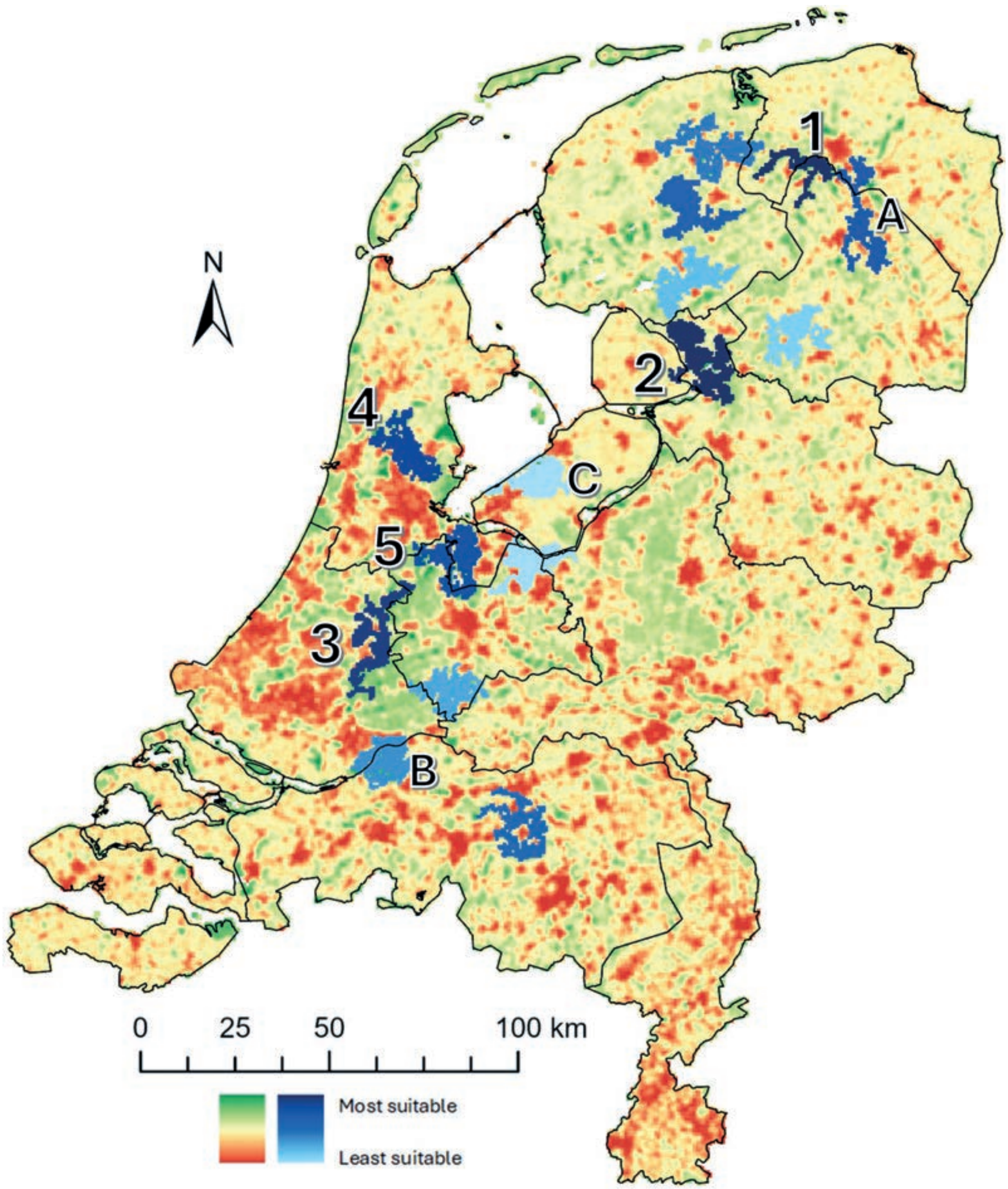

Figure 2. Habitat suitability map for the European mink in the Netherlands overlayed with the most suitable areas in blue. Shown are the top 15 contiguous areas of 50-250 km^2 with the highest suitability score. The darker blue the area, the higher the suitability score. The top five highest scoring areas are numbered and further discussed in the text, along with three other potentially interesting areas which are labelled A, B and C.

a suitability assessment, but each area comes with specific benefits and merits that are not reflected in the model. Here, we will go into detail on this. Notably, the high intensity of muskrat trapping in certain areas (Figure 3) poses a problem for a reintroduction of the European mink. The susceptibility of mink to these lethal traps is evident from the cap-

Table 3. An overview of the most suitable areas (Natura 2000 Gebieden. LVVN, retrieved on 5-9-2025)

	Area	Province(s)	Natura 2000 designation	Description
1	Onlanden and Leekstermeergebied	Groningen and Drenthe	Only included in Birds Directive	Two wetland areas, characterized by open water bodies, peat bogs and narrow canals. There is a high vegetation cover with wide reed beds.
2	Weerribben-Wieden National Park	Overijssel	Whole area is included in Habitats Directive	Two connected areas that together form one of the largest wetlands in the Netherlands with a high variety of wetland vegetation. Large parts of the area are inaccessible to the public. This area was the main release site for the reintroduction of the otter.
3	Nieuwkoopse plassen and peat meadow areas to the south	Zuid-Holland	Nieuwkoopse plassen is included in Habitats Directive	This area is similar to nr 5. In addition to lakes and many watercourses, this area is mainly characterized by extensive reed beds.
4	Lowland peat complex north of Amsterdam	Noord-Holland	Many small areas, some of which are included in the Habitats Directive.	The area is characterized by natural meadows and narrow watercourses. It includes many important meadow bird reserves.
5	Oostelijke Vechtplassen and surrounding area	Noord-Holland and Utrecht	Most of the area is included in the Habitats Directive	This area features large lakes and inaccessible peatlands. It includes the Vecht river, which flows north towards the IJsselmeer. In large parts of the area there is high recreational pressure.
Other potentially interesting areas				
A	Drentsche Aa National Park and Zuidlaardermeergebied	Drenthe and Groningen	Only Drentsche Aa included in Habitats Directive	The Drentsche Aa is a brook valley that meanders through heathlands, wetlands and forests. Nearby, the Zuidlaardermeer is a large lake surrounded by reed beds and wet meadows.
B	Biesbosch National Park	Noord-Brabant	Whole area included in Habitats Directive	A vast freshwater tidal wetland, which is formed by the confluence of the Rhine and Meuse rivers. It is a dynamic ecosystem that is almost completely forested.
C	Oostvaarders-plassen National Park	Flevoland	Only included in Birds Directive	This man-made marsh area is characterized by a high density of large herbivores. It is almost completely inaccessible for people.

tures of American mink and western polecat (on average 125 and 238 per year in the period 2007-2010) (LCCM 2011). It is therefore very likely that these traps will also pose a threat to the European mink. As already mentioned by Lange et al. (2022), the aim is to push the muskrat back to the Dutch-German border by 2034. In some regions, muskrats are effectively controlled, leading to a gradual reduction in trapping efforts. In the western lowland peat areas there is however still a very high density of muskrats and therefore also a high trapping intensity.

The Onlanden and Leekstermeergebied (1 in Figure 2) stand out as the highest-scoring region overall, making it a promising candidate for reintroduction. The area is however relatively narrow and it is only included in the Birds Directive, which makes it less suitable as currently no additional measures are obliged to be taken for the European mink after release. The Weerribben-Wieden (2) also scores very high, while also being larger and part of the Habitats Directive. Because this area served as the main release site for the reintroduction of the otter, otter-specific infrastructure has been installed across the area, like safe passages underneath roads. European mink have

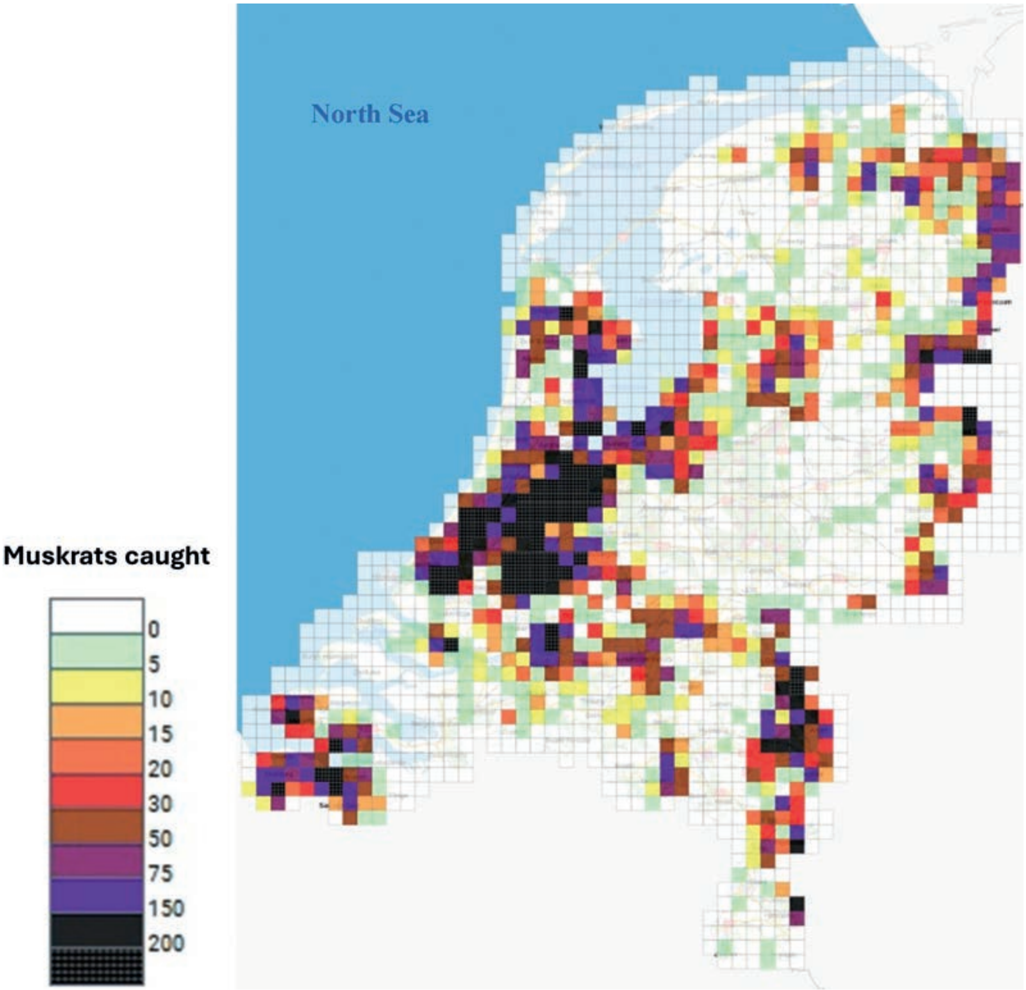

Figure 3. Total number of muskrats (*Ondatra zibethicus*) trapped per 5 x 5 km^2 in 2024. *Source*: Dolf Moerkens, Unie van Waterschappen.

also been found to use such passages (Dr. Tiit Maran, personal communication). As a result, these developments likely make the Weerribben-Wieden a particularly favourable site for a European mink reintroduction.

The Nieuwkoopse plassen (3) and Oostelijke Vechtplassen (5) both score well, although they are relatively fragmented and likely experience high human pressure from recreational activities and infrastructure such as roads. Notably, these areas suffer from a very high abundance of American crayfish. The European mink would therefore be a welcome crayfish predator here. On the other hand, these areas are also characterized by a high muskrat trapping intensity (Figure 3). Since European mink are also susceptible for these traps, these areas may currently not be a good option. However, as muskrats are being pushed back to the German border (Lange et al. 2022), the suitability of these areas may increase over time.

Another high scoring area is the lowland peat complex north of Amsterdam (4). This area however includes many important reserves for meadow birds (Sierdsema et al.

2017), which are potential prey for the European mink. Although birds constituted a small part of the European mink diet in foreign studies (Palazón et al. 2004, Sidorovich et al. 2010), it is difficult to translate this to the Dutch situation and therefore the area may be unfavourable, also because it may not provide enough vegetation cover for the European mink. Lastly, the area is very isolated, making it less suitable for a reintroduction.

Some areas, although not ranked in the top five of the model, are still worth highlighting. First of all, the Drentsche Aa and Zuidlaardermeergebied (A in Figure 2) appear very suitable, especially as it is interconnected with area 1. Another potentially suitable area is the Biesbosch (B), one of the largest and most undisturbed wetland areas in the Netherlands. However, as a river delta it is relatively polluted (Rozema et al. 2008) and the area is also surrounded by urban areas and intensive agriculture. Last of all, the Oostvaardersplassen (C) could be an interesting option. It is one of the smaller areas, but it could serve as an important connection between the northern and western wetlands in the Netherlands.

When comparing the most suitable areas in the model to the current range of the European mink, the Danube delta seems to be most resembling, as this is a wetland dominated by dense vegetation like reed beds. Other areas, including the reintroduction sites in Estonia and Germany, are generally drier, indicating that the areas in the Netherlands could be more suitable. In the feasibility study by Lange et al. (2022), it is argued that the Weerribben-Wieden is probably the best release site, with the Biesbosch and lowland peat area in Zuid-Holland as other contenders. Our results confirm the suitability of these areas, but indicate that the Biesbosch may be too isolated. Some other areas are likely more suitable, like the wetlands on the Groningen-Drenthe border.

Overall, we think the rule-based habitat suitability model was a good choice for this study. That said, there are some limitations that must be addressed. First of all, because the model was based on shoreline density, the suitability of riparian areas without many shorelines like rivers, brooks and lakes may have been underestimated. We tried to better incorporate rivers with natural floodplains by including Natura 2000 and NNN (Nature Network the Netherlands) areas into the land use map as naturally managed agricultural land. Nonetheless, rivers still seemed to be relatively unsuitable, probably due to the low shoreline length. Additionally, some meadow bird areas were classified as highly suitable because they are also categorized as naturally managed agricultural land. This classification is questionable, as these areas generally lack sufficient vegetation cover for the European mink.

Importantly, the output map shows the relative suitability (i.e. which areas are the most suitable), instead of absolute suitability (i.e. is the area suitable or not). It was impossible to create an absolute suitability map, because this requires thresholds for suitability (e.g. how many roads per km^2 do European mink tolerate), which are unknown for the European mink. This means that we cannot say for certain that the Netherlands is suitable for the European mink, but we think it is very likely. Even if the thresholds were known, there would still be uncertainty simply because there are many factors involved that determine the success or failure of a reintroduction, and it is impossible to predict these. For better suitability estimates, a more rigorous model could be used, like a Maxent model based on proxy species like American mink, western polecat and otter (Barata et al. 2024). In addition, a least-cost path analysis can be carried out to better estimate the connectivity between suitable areas.

Conclusion

According to our habitat suitability analysis, the Netherlands likely provides several areas of highly suitable habitat for the European mink. Particularly promising regions in the north include the Weerribben-Wieden and the Onlanden–Leekstermeer area, although the latter is not designated under the Habitats Directive. Due to their relative proximity to each other these areas are probably the most suitable for establishing a metapopulation, though fragmentation remains an important issue. The lowland peat areas in the west of the country with an abundance of American crayfish could also be good options. However, these areas are even more fragmented, have a high muskrat trapping intensity and a higher human presence. We recommend further investigation into the potential release sites, to (1) examine habitat characteristics in greater detail and identify area-specific ecological bottlenecks, (2) estimate the population size the areas can support, (3) assess the connectivity between areas, and (4) get a grip on the local political and societal dynamics that will play an important role in the success of the reintroduction.

Due to its long absence in the Netherlands, the European mink has become a relatively unfamiliar species. As such, it has received little attention, despite its critical conservation status. The recent disappearance of American mink in the Netherlands presents a unique opportunity to bring back a critically endangered species. This study can serve as a stepping stone by providing insights into the suitability of the Netherlands for a reintroduction of the European mink. In doing so, the study supports the research and conservation of a mammal that may sooner or later reclaim its place in the Dutch landscape. This is not only a crucial step for advancing its reintroduction in the Netherlands, but may also serve as a framework for similar reintroduction efforts in other countries.

Acknowledgements: We would like to express our gratitude to everyone from the Dutch Mammal Society who in some way contributed to this research. A big thank you to Dick Klees and Dr. Tiit Maran, whose expertise was crucial in helping develop the habitat suitability model, as well as Dolf Moerkens, for the insightful data on muskrat trapping. Lastly, we would like to thank the reviewers for their valuable feedback.

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Samenvatting

Habitatgeschiktheid voor de herintroductie van de Europese nerts (*Mustela lutreola*) in Nederland

Als een van de meest bedreigde zoogdieren in Europa heeft de Europese nerts (*Mustela lutreola*) dringend beschermingsmaatregelen nodig om de soort van uitsterven te behoeden. Een van de grootste bedreigingen is de invasieve Amerikaanse nerts (*Neogale vison*), die in Europa wijdverspreid is en hier de Europese nerts heeft verdreven. In Nederland is de Amerikaanse nerts in 2025 echter vrijwel verdwenen, wat kansen biedt voor een herintroductie van de Europese nerts. Het doel van dit onderzoek was om de meest geschikte gebieden voor een herintroductie in Nederland in kaart te brengen. Hiervoor is een *rule-based* habitatgeschiktheidsmodel ontwikkeld in de *Suitability modeler* tool in ArcGIS Pro. Het model is gebaseerd op de belangrijkste habitateisen van de Europese nerts: natuurlijk terrein

met voldoende beschutting, aanwezigheid van water, weinig wegen en afwezigheid van mensen. Volgens het model lijkt Nederland over een aanzienlijk aantal geschikte gebieden te beschikken, hoewel de geschiktheid van gebieden varieert door factoren zoals versnippering en isolatie. Ook de bestrijding van muskusratten speelt een rol, vooral in het westen van Nederland, omdat nertsen kwetsbaar zijn voor de vallen die hiervoor gebruikt worden. De moerasgebieden in Noord-Nederland komen naar voren als de meest geschikte herintroductiegebieden, met de Weerribben-Wieden en de Onlanden/Leekstermeergebied als best-scorende gebieden. Ook de Oostelijke Vechtplassen en Nieuwkoopse plassen lijken geschikt, met als bijkomend voordeel de hoge dichtheden van Amerikaanse rivierkreeft (*Procambarus clarkii* en *Procambarus acutus*), waarop de nerts kan prederen. Versnippering en intensieve muskusratbestrijding vormen hier echter nog uitdagingen. De Biesbosch, Drentsche Aa en Oostvaardersplassen zijn potentieel ook interessante gebieden, maar scoren minder hoog in het model. Vervolgonderzoek wordt aanbevolen om (1) gebiedseigenschappen verder in kaart te brengen, (2) connectiviteit tussen gebieden te bepalen en (3) inzicht te krijgen in de lokale politieke en maatschappelijke factoren die belangrijk zijn voor een succesvolle herintroductie. Voor zover wij weten, is dit de eerste habitatgeschiktheidsstudie voor de Europese nerts in Europa. De bevindingen kunnen niet alleen bijdragen aan de ontwikkeling van een herintroductieplan in Nederland, maar ook als voorbeeld dienen voor vergelijkbare studies in andere landen.

Received: 1 March 2025
Accepted: 11 April 2025

The closure of mink farms in the Netherlands: a unique opportunity for European mink

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Keywords: American mink, exotic species, reintroduction, population, Sars-CoV-2.

Abstract: European mink (*Mustela lutreola*) is a medium-size mustelid and highly endangered in Europe. The species was historically widespread in continental Europe, but the species has suffered a serious decline since the middle of the 19th Century. A major threat for the remaining European mink populations in Europe are escaped American mink (*Neogale vison*), which outcompete European mink in the wild. However, the farming of American mink was banned in the Netherlands during the global outbreak of Sars-CoV-2. Afterwards, the by-catch of American mink in the National Programme of Muskrat Trapping significantly decreased, demonstrating that American mink observed or trapped in the Netherlands mainly escaped from mink farms. The virtual disappearance of American mink in the wild offers a unique opportunity to reintroduce the highly endangered European mink in the Netherlands.

Introduction

American mink has been brought to Europe for fur farming, but individuals regularly escaped from these farms and were capable of surviving in the wild (Genovesi et al. 2009, Dekker & Hofmeester 2014). In several European countries these escapes have resulted in large and increasing populations of American mink. In contrast to other European countries American mink never established a viable population in the Netherlands, despite a wide range of suitable habitat and a continous influx of escaped minks (Dekker & Hofmeester 2014, Vada et al. 2023).

The American mink is a threat for the last remaining and highly endangered populations of European mink as it outcompetes the latter. Hence, in areas free of American mink it should theoretically be possible to (re)introduce European mink. The closure of mink farms in the Netherlands in 2021 eliminated an important threat and offers a unique opportunity to succesfully reintroduce the highly endangered European mink.

The closure of mink farms

In the Netherlands the continuous flow of escaping American mink abruptly stopped in 2021 during the global outbreak of Sars-CoV-2, the virus that causes Covid-19 in humans, when all remaining mink farms in the Netherlands were closed. American mink proved to be extremely susceptible to the virus and mink on several dozens of mink farms in the south of the Netherlands became infected (Sikkema et al. 2022). The government subsequently decided to close all mink farms to prevent emergence of new virus variants on these farms. In the course of 2021, all mink farms in

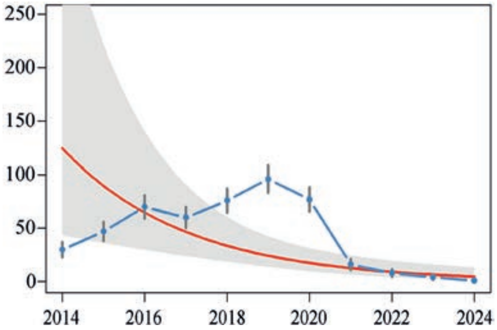

Figure 1. The number of catches of American mink per year (blue line) as by-catch of muskrat control 2014-2024. The red line is the trend calculated by TRIM over this period (calculation performed by Bing Verbrugh). Over the same period, brown rat by-catch was stable (data not shown). Data made available by the Dutch Water authorities.

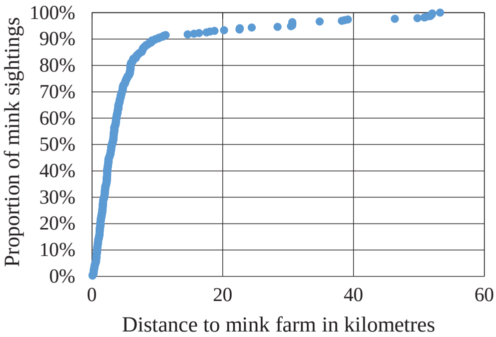

Figure 3. The distance between a sighting of a mink and the nearest mink farm in the period 2015-2020. 90% of all sightings were within 10 km of a mink farm. Data from the NDFF.

the Netherlands were closed, ending the commercial farming of American mink in the Netherlands. An unintended effect of closing all mink farms was that it stopped the continuous ‘introduction’ of captive mink to the wild. A statistical analysis of by-catch per year of muskrat trappers in the Netherlands, based on data from the Dutch Water authorities, shows a strong decreasing trend of trapped minks as by-catch in the period 2014-2024, with a clear trend break in 2021 (Figure 1), the year in which the mink farms were closed (Figure 1).

Most recently, in 2025, the number of validated sightings of American mink that are recorded in the Netherlands and deposited in the National Databank Flora and Fauna (NDFF) is less than five confirmed records per year.

Correlation between farms and sightings

Dekker & Hofmeester (2014) had already provided evidence that in the Netherlands most American mink in the wild come directly from farms and this was, once again, confirmed in 2021 (Sikkema et al. 2022). This article revealed a clear correlation between the location of farms and the number of validated sightings (mostly by-catch of muskrat trapping) of American mink within blocks of 5x5 km during the period 2015-2020 (Figure 2). The map shows an obvious correlation between the presence of a (former) mink farm and the presence of this species in the wild. Also the calculated distance from a sighting to the nearest farm, shows that 90% of the sightings or by-catch of American mink were made within 10 km of a farm and, moreover, a part of the sightings at a greater distance are linked to farms that were closed longer ago or sighted mink came from farms in neighbouring countries (Figure 3).

Although American mink are capable of surviving in the wild in the Netherlands (Genovesi et al. 2009, Dekker & Hofmeester 2014, Bouwens 2017), the species never established a viable free-ranging population, in contrast with other European countries (Vada et al. 2023). As the American mink has no natural predators in the Netherlands and given the wealth of suitable habitat, it is presumed that the intensive control of muskrat (*Ondatra zibethicus*) has played a role in preventing American mink to establish. American mink is regularly registered as an unintentional by-catch in the trapping devices of muskrat trappers. The Dutch Water authorities register all by-catch and most records of American mink comes from their database.

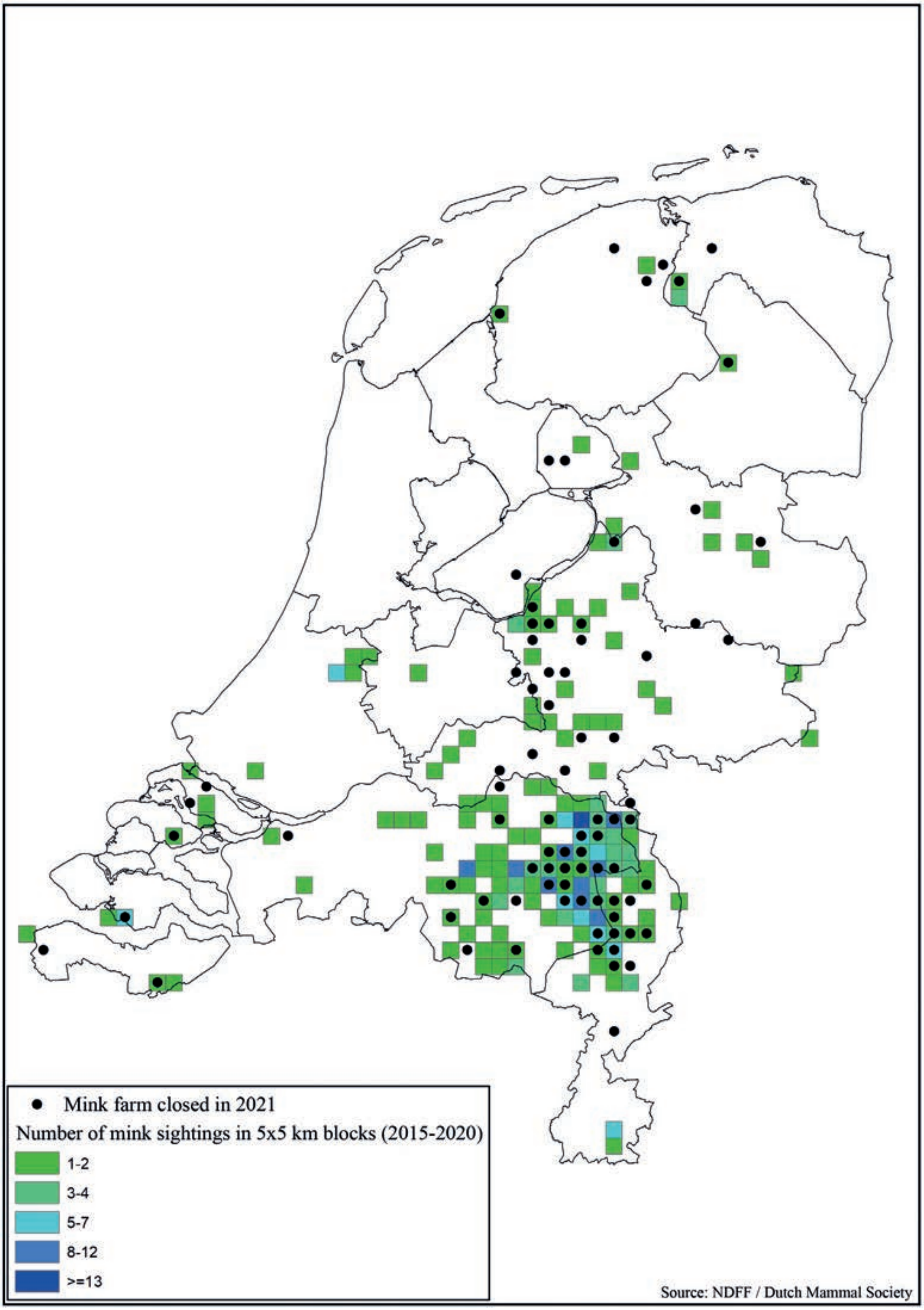

Figure 2. The location of mink farms (black dots) and the number of validated sightings of American mink within blocks of 5x5 km during the period 2015-2020. Data from the NDFF (National Databank for Flora and Fauna). Map compiled by Martijn van Oene, Dutch Mammal Society.

Discussion

With the closing of mink farms in the Netherlands the introduction of American mink into the wild has stopped and thanks to the Muskrat Trapping Programme the last surviving individuals will probably end up as bycatch. It is therefore expected that this species will finally disappear from the Netherlands within the next few years, although it will be still possible that individuals from neighboring countries will migrate into the Netherlands.

The semi- or final disappearance of the American mink offers a unique opportunity for the 'critically endangered' European mink (*Mustela lutreola*). Only a few relict populations of this species are present in southern and eastern Europe. The strong decline of this species is not fully understood (Maran & Henttonen 1995), but it is clear that American mink is an important threat for European mink through direct competition and interference competition (Maran 2007, Santulli Sanzo et al. 2014), which hinder the recovery of the species.

The absence of American mink in the Netherlands, literally creates a unique opportunity for the conservation and recovery of European mink. Exploratory studies of the possibilities for reintroducing this species in the Netherlands (Lange 2022, Zwartenkot 2024, Zwartenkot et al. 2025), show that several large marsh areas in west and northern parts of the Netherlands are suitable locations for reintroduction.

However, the trapping of muskrats is not only a danger for American mink, it is also a threat for European mink. Fortunately, in the Netherlands a strategy has been adopted to 'eliminate the muskrat to the state border'. This strategy has been evaluated and it shows that it is a feasible and rational strategy (Bos & Gronouwe 2018). This means that in the coming years large areas in the Netherlands will be more or less muskrat free and that the number of traps in these areas will decrease significantly, also reducing the number of bycatch and the risk for European mink being trapped. When detecting muskrats, the use of 'smart traps', which only close when the target species is trapped (see website: https://lifemica.nl/research-innovaties/smart-life-traps/), will prevent the trapping of non-target specimens. The closing of mink farms in the Netherlands therefore has provided a unique opportunity for the critically endangered European mink.

Acknowledgements: Henry Kuipers and Daan Bos (Van Hall Larenstein, University of applied sciences) and the Dutch Water authorities are thanked for their help in doing the statistical analysis (HK & DB) and providing access to the database of bycatches.

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Samenvatting

Het sluiten van de nertsenfarms in Nederlands biedt een unieke kans voor de Europese nerts

De Europese nerts (*Mustela lutreola*) is een middelgrote marter en ernstig bedreigd in Europa. Historisch was de soort veel wijder verspreid, maar de verspreiding is sterk gekrompen in de 19e eeuw. Een belangrijke bedreiging voor de laatste populaties van Europese nerts zijn ontsnapte Amerikaanse nertsen (*Neogale vison*), die in het wild de Europese nerts verdringen. Tijdens de Sars-CoV-2 pandemie zijn de nertsenfarms echter gesloten. Nadien is het aantal bijvangsten van Amerikaanse nerts door muskusrattenvangers significant gedaald, wat aantoont dat de meeste Amerikaanse nertsen in Nederland ontsnappingen waren uit nertsenfarms. Het verdwijnen van de Amerikaanse nerts biedt daarmee een unieke kans om de ernstig bedreigde Europese nerts in Nederland te herintroduceren.

Received: 15 April 2025
Accepted: 4 November 2025


ED.

First exploration of currently used pesticides in garden dormouse in the Netherlands

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Abstract: The garden dormouse (*Eliomys quercinus*) is classified as *Critically Endangered* in the Netherlands. This study investigated the exposure of garden dormice to currently used pesticides in the Netherlands by analyzing pesticide residues in brain, fur, liver and stomach tissues of five individuals. In this first exploratory assessment for garden dormouse in the Netherlands, thirteen compounds were detected, including insecticides and herbicides. The study was limited by its small sample size (n=5) and variable sample quality. Consequently, it provides a first indication of the exposure of the Dutch garden dormouse population to currently used pesticides, but no conclusions can yet be drawn regarding the extent, severity, or ecological significance of this exposure.

Keywords: garden dormouse (*Eliomys quercinus*), pesticide exposure, currently used pesticides, wildlife toxicology, small mammals, herbicides, insecticides, critically endangered, hibernation, the Netherlands.

Introduction

The garden dormouse (*Eliomys quercinus*) is classified as *Critically Endangered* on the current Dutch Red List (van Norren et al. 2020) and has undergone a sharp population decline across its European range, resulting in a *Vulnerable* status on the Global Red List (Bertolino et al. 2024).

The Dutch garden dormouse population occurs only in the southern part of the province of Limburg. The two remaining populations live in the Savelsbos and Bemelerberg nature reserves and inhabit a fragmented landscape close to agricultural and urban areas (Norren et al. 2024). In the Netherlands, a garden dormouse conservation project has been ongoing since 2018 (Feys & Nijs 2018), focusing mainly on habitat restoration, communication and research. Recent research comprised food web and habitat use (Nijssen & Hiddes 2020, van Norren & Schepers 2023), genetic diversity (La Haye 2019) and population monitoring (van Norren et al. 2023, 2024).

Research on pesticides in garden dormice has not yet been conducted in the Netherlands, but such studies may be relevant because large-scale research has demonstrated negative sublethal effects of pesticides on non-target organisms (Wan et al. 2025). A German study suggested that pesticide exposure also applies to garden dormouse populations (Famira-Parcsetich et al. 2022, Büchner et al. 2024). The proximity of garden dormice to both agricultural and urban areas makes

Figure 1. The six garden dormice (*Eliomys quercinus*) that were analyzed. *Photo: Aafke Saarloos.*

the species potentially susceptible to pesticide exposure from various sources, such as households and agriculture. The garden dormouse mainly forages on arthropods such as insects, millipedes and spiders living in the litter layer and vegetation, and also consumes plant-based food sources, such as flowers, berries and other plant parts (Storch 1978). As a considerable part of their diet includes animal prey, garden dormice may be susceptible to the bioaccumulation of environmental contaminants. During torpor or hibernation, the lipid-metabolism system may result in release of chemicals stored in the lipids, which may make the garden dormouse vulnerable to lipophilic compounds.

The present study provides the first report on exposure to currently used pesticides in the Dutch garden dormouse population. Pesticide residues were analyzed in liver, fur, brain, and stomach tissues of five individuals of garden dormouse to assess the extent of exposure.

Material and methods

Sample collection and study area

Between 2015 and 2023, five carcasses of garden dormouse (*Eliomys quercinus*) were found in the Savelsbos and Bemelerberg nature reserves, located in the south of the province of Limburg, the Netherlands (Figure 1). All carcasses were collected opportunistically after natural death, therefore no permits or ethical approvals under Dutch animal welfare legislation were needed. The transport of the carcasses was carried out under permit. The carcasses were stored at -18 °C.

Dissection and tissue sampling

Each animal was weighed and measured (length, from head to tail tip) upon dissection. Observations were recorded on the state of decay, the presence or absence of target

Table 1. Individual characteristics of the garden dormouse carcasses included in the pesticide analysis. Length is head to end of tail.

#	Weight (g)	Length (cm)	Maggots	Area and date of collection	State of tissue
1	29.0	X*	Yes (a lot)	Bemelerberg 06-09-2022	Excluded from analysis Very bad, almost no tissue left, either eaten by maggots or other necrophagous animals; very advanced state of decay.
2	X	25.0	No	Bemelerberg 13-07-2022	Good
3	X*	16.5	Yes	Savelsbos 05-06-2015	Rather bad, advanced state of decay, organs were small and dry
4	65.2	24.5	No	Bemelerberg (Mettenberg) 01-09-2022	Good (Figure 2a)
5	66.5	23.5	No	Savelsbos or Bemelerberg, no date	Bad, Internal organs not present, potentially eaten by necrophagous animals (Figure 2b)
6	51.8	24.0	Yes	Bemelerberg (Koelebosch) 08-09-2023	Rather bad, Integrity of internal organs affected by maggots (Figure 2c)

X*: Not possible to measure, because of the advanced state of decay.

organs and the occurrence of necrophagous insects such as maggots.

Extraction procedure

Tissue extraction followed a slightly modified QuEChERs method (Anastassiades et al. 2003). Briefly, tissue samples were weighed and homogenized in acetonitrile (ACN). Ultrapure Milli-Q water was added, and samples were agitated using a head-over-head shaker for three hours. A salt mixture containing magnesium sulfate ($MgSO_4$), sodium chloride (NaCl), trisodium citrate 5,5-hydrate ($C_6H_5Na_3O_7\cdot 5.5H_2O$) and di-sodium hydrogen citrate 1,5-hydrate ($C_6H_6Na_2O_7\cdot 1.5H_2O$) was added to the solution. Per 5 g of tissue, the salt mixture contained 4 g MgSO4, 1g NaCl2, 1 g $Na_3C_6H_5O_7\cdot 2H_2O$, 0.5 g $Na_2HC_6H_5O_7\cdot 5H_2O$. After an additional three hours of shaking, samples were sonicated with Bandelin Sonorex RK100 and centrifuged in a Sigma 2-16KL centrifuge at 5000 rpm for five minutes. Both ACN and water layers were recovered and stored at -80 °C until further analysis.

Pesticide identification and quantification

Extracts were analyzed using a Shimadzu LC-MS/MS 8045 system with a Shim-pack Velox Biphenyl column (2.1 mm x 100 mm, 2.7 µm). The column oven temperature was maintained at 35 °C, injection volume 1 µL and a flowrate of 0.4 mL/min. The mobile phases consisted of 2 mmol/L ammonium formate + 0.002% formic acid in ultrapure Milli-Q water (mobile phase A) and 2 mmol/L ammonium formate + 0.002% formic acid in methanol (mobile phase B).

The samples were first screened qualitatively for the presence of 648 pesticides (Appendix). The presence of compounds with preliminary detection signals were confirmed using certified analytical standards (PESTANAL®). Pesticide concentrations were determined by comparing sample

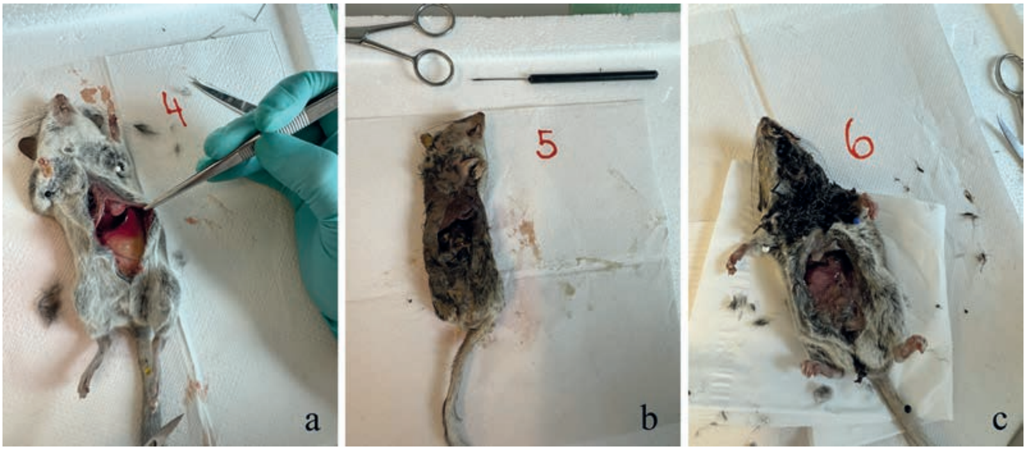

Figure 2. Photos illustrating variation in tissue condition across individuals: a. Individual #4, showing a relative big and yellowish colour of the stomach. b. Individual #5, showing the absence of internal organs. c. Individual #6, showing a dark colour of the liver. *Photos: Aafke Saarloos.*

Table 2. Linearity (r^2), accuracy range and limit of quantification (LOQ) for each of the 13 pesticides detected in at least one sample.

Pesticide	Linearity (r^2)	Accuracy (%)	LOQ (µM)
3-Indolyl-acetic acid	0.999	90.1-103.3	0.05
Mesotrione	0.999	91.6-106.9	0.01
Chloridazon	0.999	85.6-114.5	0.005
Deet	0.999	90.9-106.8	0.01
Fluopyram	0.999	97.9-104.5	0.005
Spirotetramat	0.999	95.6-107.4	0.01
Quinalphos	0.999	76.1-103.9	0.01
Isoxadifen-ethyl	0.999	86.4-115.4	0.005
Terbufos	0.999	90.6-105.8	0.05
Fenoxaprop-ethyl	0.999	93.3-105	0.05
Etoxazole	0.999	95.1-107.9	0.01
Resmethrin	0.998	95.9-109.3	0.005
Permethrin	0.999	94-112.8	0.05

signals with standard curves generated from these reference compounds.

All calibration curves showed excellent linearity, with r^2 values above 0.99 (Table 2). The limit of quantification (LOQ) for each compound was determined based on the lowest concentration showing recovery between 75% and 125%. Accuracy values for the 13 detected pesticides ranged from 76% to 115%.

Results

Decomposition and sample integrity

Considerable variation in the state of decomposition was observed among the five garden dormouse carcasses (Table 1). The five individuals were in different states at the moment of collection; two individuals (#3 and #6) showed advanced decay with partial loss of internal organs, one individual (#5) had no

Table 3. List of pesticides detected in at least one tissue sample (individual #1 was not analyzed).

Pesticide	Type	# individuals detected
3-Indolyl-acetic acid	Plant hormone, (also) occurring naturally	5 out of 5
Chloridazon	Herbicide	1 out of 5
DEET	Insect repellent	2 out of 5
Etoxazole	Insecticide	4 out of 5
Fenoxaprop-ethyl	Herbicide	1 out of 5
Fluopyram	Fungicide / nematicide	2 out of 5
Isoxadifen-ethyl	Herbicide antidote	5 out of 5
Mesotrione	Herbicide	2 out of 5
Permethrin	Insecticide	1 out of 5
Quinalphos	Insecticide / acaricide	3 out of 5
Resmethrin	Insecticide	5 out of 5
Spirotetramat	Insecticide	2 out of 5
Terbufos	Insecticide / nematicide	5 out of 5

Table 4. Pesticide concentrations (ng/g) in liver, fur, brain and stomach of garden dormouse #2 - #6. ND (green) means compound not determined in tissue. NA (grey) means tissue was not available. LOQ = Limit of Quantification (the pesticide has been found in the tissue, but the amount is too small to quantify).

Individual #	2	2	2	2	3	3	3	3	4	4	4	4	5	5	5	5	6	6	6	6
Tissue from	Liver	Fur	Brain	Stom.	Liver	Fur	Brain	Stom	Liver	Fur	Brain	Stom	Liver	Fur	Brain	Stom	Liver	Fur	Brain	Stom.
3-Indolyl-acetic acid	70	1747	37	68	134	10,507	ND	127	89	17,697	10	83	NA	350	73	NA	114	1647	NA	142
Chloridazon	ND	ND	ND	ND	ND	ND	ND	ND	ND	<LOQ	ND	ND	NA	ND	ND	NA	ND	ND	NA	ND
DEET	ND	<LOQ	ND	ND	ND	505	ND	<LOQ	ND	ND	ND	ND	NA	ND	ND	NA	ND	ND	NA	ND
Etoxazole	<LOQ	ND	ND	ND	ND	ND	ND	ND	<LOQ	ND	ND	ND	NA	<LOQ	ND	NA	ND	<LOQ	NA	ND
Fenoxaprop-ethyl	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	NA	ND	ND	NA	ND	ND	NA	1187
Fluopyram	ND	<LOQ	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	NA	ND	ND	NA	ND	<LOQ	NA	ND
Isoxadifen-ethyl	ND	24	ND	ND	106	30	ND	5	ND	<LOQ	ND	4	NA	257	ND	NA	4	18	NA	ND
Mesotrione	8	ND	ND	ND	ND	ND	ND	ND	<LOQ	ND	ND	ND	NA	ND	ND	NA	ND	ND	NA	ND
Permethrin	ND	ND	ND	ND	ND	ND	ND	ND	ND	<LOQ	ND	ND	NA	ND	ND	NA	ND	ND	NA	ND
Quinalphos	ND	ND	ND	ND	<LOQ	<LOQ	ND	<LOQ	ND	ND	ND	<LOQ	NA	ND	ND	NA	<LOQ	<LOQ	NA	13
Resmethrin	<LOQ	<LOQ	<LOQ	7	ND	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	5	NA	<LOQ	<LOQ	NA	ND	17	NA	ND
Spirotetramat	ND	ND	ND	ND	66	ND	ND	38	ND	ND	ND	ND	NA	ND	ND	NA	69	ND	NA	63
Terbufos	ND	149	<LOQ	99	ND	149	<LOQ	36	ND	<LOQ	ND	ND	NA	<LOQ	<LOQ	NA	43	173	NA	187
# pesticides	4	6	3	3	4	6	2	7	4	6	2	4		5	3		5	7		5

usable liver or stomach tissue due to scavenger activity, and one individual (#6) lacked usable brain tissue (Figures 2a-c).

Pesticide analysis

A total of 13 currently used pesticides were detected in different tissues (Table 3 and 4). The identified compounds included:

- Herbicides: chloridazon, fenoxaprop-ethyl, mesotrione
- Insecticides: etoxazole, permethrin, quinalphos, resmethrin, spirotetramat, terbufos
- Fungicide: fluopyram
- Insect repellent: DEET
- Herbicide antidote: isoxadifen-ethyl
- Plant hormone: 3-indolyl-acetic acid

The prevalence and concentrations of compounds varied among individuals, as did the tissue types available for analysis. Results per individual, including compound concentrations per tissue (liver, fur, brain and stomach), are presented in Table 4. The plant hormone 3-indolyl acetic acid was detected in nearly all samples, whereas the other compounds were present only in part of the tissues. The detection of this naturally occurring compound should be interpreted with caution as 3-indolyl acetic acid is a natural chemical, commonly found in nearly all environmental samples analyzed.

A substantial proportion of detected compounds were present at concentrations below the limit of quantification (LOQ), indicating low levels that cannot be reliably quantified. While the presence of these substances was confirmed, their exact concentrations remain undetermined (Table 4).

Discussion

This study provides the first exploratory investigation of currently used pesticide residues in the critically endangered garden dormouse population in the Netherlands. The detection of 13 pesticide residues in tissues from five individuals from two different locations confirms that garden dormice are exposed to such compounds in South Limburg, the area where the remaining Dutch populations occur.

Although pesticide compounds were detected in multiple tissues, these findings should be interpreted with caution, due to limitations in sample size and sample quality. The small number of carcasses analyzed (n=5), combined with a variable and often advanced state of decomposition, severely constrained the availability and integrity of organs suitable for chemical analysis. The decomposed state of the tissues may have resulted in degradation of chemicals, potentially underestimating exposures. Consequently, no firm conclusions can be drawn regarding the extent or ecological significance of the exposure to currently used pesticide in this population.

Multiple compounds were detected at low concentrations; however, their potential combined effects on garden dormice remain unknown. Furthermore, we don't know the exposure pathways (Buijs & Mantingh 2022) of pesticides for the garden dormice, because these have not been determined in a way Buijs & Mantingh (2022) have done for other species. It is unclear whether exposure occurs via contact with contaminated substrates, ingestion during grooming, or dietary intake of contaminated food items such as berries, nuts, or insects.

The plant hormone 3-indolylacetic acid, detected in nearly all samples, is found in most environmental samples, be it soil, vegetation or biota. Its presence in our samples is difficult to interpret, as it occurs both as a result of pesticide use and also as a naturally occurring plant hormone in the environment. Therefore, it is not clear what the background of this signal is.

Other currently used compounds such as glyphosate-based herbicides were not included, as these substances are difficult to be reliably measured in tissues. PFAS is another group that may be of interest, however, the methods applied in the current study did not allow their detection. Older pesticides that are no longer in use, for instance DDT and Dieldrin, may still be found in the garden dormice; however, they were not included in this study because they provide limited practical management perspective.

Given the limited availability of garden dormouse carcasses in the Netherlands, opportunities for further research are restricted. Nevertheless, future studies could aim to:

1. Increase sample size (i.e. increase the number of collected carcasses), or, if this is not feasible, collect blood samples from living individuals. Blood reflects the compounds circulating in the organism at the time of sampling and can therefore provide informa-

tion on recent pesticide exposure. 2. Collect and analyze droppings, which can indicate metabolic processes and the excretion of various compounds. 3. Include a broader range of contaminants, such as PFAS, glyphosate-based herbicides and legacy pollutants (PCBs, DDT, Dieldrin). 4. Investigate potential exposure pathways by analyzing pesticides in soil, food items (berries, fruits), and prey (insects). 5. Explore sublethal effects including immune suppression, reproductive impairments and disruptions to hibernation; and 6. Compare pesticide exposure with other mammal species (e.g. bats, hedgehogs) to better understand food web contamination.

Collaborations with transboundary research initiatives (e.g. German or Belgian studies) may be essential for obtaining more robust insights, given the limited and fragmented Dutch population.

Although this exploratory study provides a first indication that garden dormice in the Netherlands are exposed to multiple pesticide compounds, further research is required before any conclusions can be drawn regarding the role of pesticides in the species' decline in the Netherlands.

Acknowledgements: We thank all supporters in Savelsbos and Bemelen, including the Vogelwerkgroep Bemelen and students participating in garden dormouse research, for collecting carcasses. We are also thankful to the German researchers of *Spurensuchen Gartenschläfer*, particularly Sven Büchner and Johannes Lang, for their generous cooperation. We also thank the reviewers for their valuable time and constructive feedback, which helped improve this article.

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Samenvatting

Een eerste verkenning van de blootstelling van eikelmuizen (*Eliomys quercinus*) aan momenteel gebruikte pesticiden in Nederland

De eikelmuis (*Eliomys quercinus*) is in Nederland geclassificeerd als Ernstig Bedreigd. In deze studie is voor het eerst de blootstelling van eikelmuizen in Nederland aan momenteel gebruikte pesticiden onderzocht. De hersenen, de vacht, de lever en maagweefsel van vijf individuen werden geanalyseerd. Dertien pesticiden werden gedetecteerd, waaronder insecticiden en herbiciden. De studie werd beperkt door het kleine aantal onderzochte dieren (*n*=5), en de variabele kwaliteit van de monsters. Hierdoor vormt het onderzoek een eerste indicatie over de blootstelling van de Nederlandse eikelmuispopulatie aan momenteel gebruikte pesticiden, maar kunnen er nog geen conclusies worden getrokken over de omvang, ernst of ecologische betekenis van deze blootstelling.

Received: 1 October 2025
Accepted: 6 November 2025

Appendix

List of compounds the garden dormouse tissue samples were screened for.

#	Compound	#	Compound	#	Compound
1	Methamidophos	217	Pyroquilon	433	Cycloate
2	Cyromazine	218	Fenthion-sulfoxide	434	Phorate
3	Acephate	219	Simeconazole	435	Propiconazole (stereo isomer)
4	Nicotine	220	Methoprotryne	436	Fenothiocarb
5	Hymexazol	221	Azobenzene	437	Mefenacet
6	Picloram	222	Iodosulfuron-methyl	438	Pyraflufen-ethyl
7	Propamocarb	223	Mexacarbate	439	Triflumizole
8	Omethoate	224	Difenoxuron	440	Phenthoate
9	Butocarboxim-sulfoxide	225	Quizalofop (free acid)	441	Cyflufenamid
10	Dinotefuran	226	Quizalofop-P	442	Thiobencarb
11	Aldicarb-sulfoxide	227	Triasulfuron	443	Oxadiargyl
12	Asulam	228	Dodine	444	Thenylchlor
13	Dicyclanil	229	Disulfoton-sulfone	445	Azoxystrobin
14	Butocarboxim-sulfone	230	Metazachlor	446	Diflufenican
15	Aldicarb-sulfone (Aldoxycarb)	231	Methiocarb	447	Cafenstrole
16	Flonicamid	232	Flurtamone	448	Bromfenvinfos
17	Simazine-2-hydroxy	233	Monalide	449	Edifenphos
18	Atrazine-desisopropyl	234	Halosulfuron-methyl	450	Orbencarb
19	Monocrotophos	235	Ethametsulfuron-methyl	451	Pirimiphos-methyl
20	Oxamyl	236	Pyrimethanil	452	Di-allate
21	Fluroxypyr	237	Aziprotryne	453	Benzyldimethyltetradecyl-ammonium Chloride
22	Oxydemeton-methyl	238	Benoxacor	454	Benzoylprop-ethyl
23	Methomyl	239	Foramsulfuron	455	Fenthion
24	Nitenpyram	240	Metalaxyl	456	Haloxyfop-P-methyl
25	2,6-Dichlorobenzamide	241	Metalaxyl-M (Mefenoxam)	457	Haloxyfop-methyl
26	Clothianidin	242	Dodemorph	458	Indanofan
27	Demeton-S-methyl-sulfone	243	Demeton-O	459	Didecyldimethylammonium
28	Atrazine-2-hydroxy	244	Demeton-S	460	Kresoxim-methyl
29	Quinclorac	245	Nuarimol	461	Disulfoton
30	Thiofanox-sulfoxide	246	Fluxapyroxad	462	Bromuconazole
31	Dicrotophos	247	Iprovalicarb	463	Clodinafop-propargyl
32	3-Indolyl-acetic acid	248	Fluopyram	464	Dinocap
33	Atrazine-desethyl	249	Flutolanil	465	2,4,6-Tribromophenol
34	Fenuron	250	Trifloxysulfuron	466	Pencycuron
35	Quinmerac	251	Trinexapac-ethyl	467	Dithiopyr
36	Mesotrione	252	Diphenylamine	468	Tolclofos-methyl
37	Pymetrozine	253	Azaconazole	469	Butamifos
38	Thiamethoxam	254	Prometryn	470	Triphenyl phosphate
39	Ethiofencarb-sulfoxide	255	Terbutryn	471	Prallethrin
40	Carbofuran-3-hydroxy (3-Hydroxycarbofuran)	256	Metominostrobin (E, Z)	472	Benalaxyl
41	Ethiofencarb-sulfone	257	Thifluzamide	473	Tebupirimfos

42 Thiofanox-sulfone
43 Carbendazim
44 Dimethoate
45 Isocarbamid
46 Ethidimuron
47 Chloridazon
48 Vamidothion
49 Dioxacarb
50 Cymoxanil
51 Methiocarb-sulfoxide
52 Mefluidide
53 Terbuthylazine-2-hydroxy
54 6-chloro-3-phenylpyridazin-4-ol
55 Sebuthylazine-desethyl
56 Thiazafluron
57 3-(3-Indolyl)-propionic acid
58 Mevinphos
59 Carbetamide
60 Imidacloprid
61 Fuberidazole
62 Aminocarb
63 Aldicarb
64 Oxycarboxin
65 Monuron
66 Methiocarb-sulfone
67 Fenthion-oxon-sulfoxide
68 Metolcarb (MTMC)
69 N-(2,4-Dimethyl-phenyl)formamide
70 Thiabendazole
71 1-(3,4-Dichlorophenyl)urea
72 Thidiazuron
73 Metoxuron
74 Imazaquin
75 Terbuthylazine-desethyl
76 Dimetilan
77 Fenthion-oxon-sulfone
78 Propoxycarbazone-sodium
79 Dichlorvos
80 Carbofuran-3-keto
81 Terbumeton-desethyl
82 Simazine
83 Cyanazine
84 Fluometuron
85 Sulfaquinoxaline
258 Bromobutide
259 Azimsulfuron
260 Barban
261 Cyproconazole
262 Clomazone
263 Fensulfothion
264 Saflufenacil
265 Oxasulfuron
266 Fenthion-oxon
267 Nitrothal-isopropyl
268 Rimsulfuron
269 Chlorantraniliprole
270 Valifenalate
271 Neburon
272 Thiophanate-ethyl
273 Fenhexamid
274 Warfarin
275 Dimethachlor
276 Triapenthenol
277 Benthiavalicarb-isopropyl
278 Bispyribac-sodium
279 Pyroxsulam
280 Isofenphos-oxon
281 Isoxaflutole
282 Propetamphos
283 Ofurace
284 Diphenamid
285 Triadimefon
286 Trietazine
287 Sulfosulfuron
288 Climbazole
289 Diclosulam
290 Famphur
291 Anilazine
292 Fensulfothion-sulfone
293 Myclobutanil
294 Tetraconazole
295 Primisulfuron-methyl
296 Mepronil
297 Chloroxuron
298 Diclobutrazol (stereo isomer)
299 Ethofumesate
300 Terbufos-sulfoxide
301 Fenamidone
474 Metaflumizone
475 Mefenpyr-diethyl
476 Tebufenpyrad
477 Pyraclofos
478 Anilofos
479 Phoxim
480 Isoxadifen-ethyl
481 Prosulfocarb
482 MCPA-butoxyethyl ester
483 Clethodim (isomer)
484 Isoxathion
485 Terbufos
486 Famoxadone
487 Profenofos
488 Cyanofenphos
489 Azinphos-ethyl
490 Chlorpyrifos-methyl
491 Sethoxydim (isomer)
492 Clomeprop
493 Dichlofenthion
494 Fluazifop-P-butyl
495 Fluazifop-butyl
496 Fluacrypyrim
497 Prochloraz
498 Oxadiazon
499 Allethrin
500 Fenchlorazol-ethyl
501 Fenthion-sulfone
502 Clofentezine
503 Dimepiperate
504 Haloxyfop-2-ethoxyethyl
505 Trifloxystrobin
506 Pretilachlor
507 Fluroxypyr-1-methylhepty-lester
508 Fluoxastrobin
509 Tri-allate
510 Buprofezin
511 Esprocarb
512 Picolinafen
513 Tralkoxydim
514 Benzyldimethylhexadecyl-ammonium Chloride
515 Aramite
516 Pirimiphos-ethyl
517 Metrafenone

86 Bromacil
87 1-naphthaleneacetamide
88 Metamitron
89 Allidochlor
90 Acetamiprid
91 1-(4-Isopropylphenyl)urea
92 Tetraethylpyrophosphate
93 Pirimicarb-desmethyl
94 Benzthiazuron
95 4-(3-Indolyl)-butyric acid
96 Fensulfothion-oxon
97 Ethirimol
98 1-(3,4-Dichlorophenyl)-3-methylurea
99 Cycloheximide
100 Karbutilate
101 Tritosulfuron
102 Tebuthiuron
103 Propoxur
104 Phosphamidon
105 Naptalam
106 Fensulfothion-oxon-sulfone
107 Fluazifop
108 Fluazifop-P (free acid)
109 1-(4-Isopropylphenyl)-3-methylurea
110 Dimethirimol
111 Monolinuron
112 Metribuzin
113 Tolylfluanid Metabolite (DMST)
114 Paraoxon-methyl
115 Imazamethabenz-methyl (isomer)
116 Atraton
117 Chlorotoluron
118 Propham
119 Cymiazole
120 Phosfolan
121 Benzanilide
122 Dichlormid
123 Forchlorfenuron
124 Atrazine-d5
125 1,3-Diphenyl urea
126 Atrazine
127 Diuron-d6
128 Difenzoquat (Difenzoquat-methyl-sulfate)
302 Silthiofam
303 Flazasulfuron
304 Triticonazole
305 Fluopicolide
306 EPTC
307 Cloransulam-methyl
308 Boscalid
309 Fenpropimorph
310 Crotoxyphos
311 Beflubutamid
312 Molinate
313 Crufomate
314 Terbufos-sulfone
315 Imazosulfuron
316 Irgarol 1051
317 Methoxyfenozide
318 (Z)-Ferimzone
319 Dimethenamid
320 Dimethametryn
321 Dipropetryn
322 Flupyrsulfuron-methyl
323 Carpropamid
324 Penoxsulam
325 Imazalil
326 Fenarimol
327 Tebuconazole
328 Hexaconazole
329 Methidathion
330 Flurochloridone
331 Metosulam
332 Prothioconazole-desthio
333 Coumachlor
334 Flubendiamide
335 Fluridone
336 Furalaxyl
337 Diflubenzuron
338 Fenamiphos
339 Thiodicarb
340 Mepanipyrim
341 Benthiazole
342 Tebutam
343 Mesosulfuron-methyl
344 Diniconazole
518 Butachlor
519 Difenacoum
520 Pyrazoxyfen
521 Alanycarb
522 Difenoconazole (isomer)
523 Cycloxydim
524 Flocoumafen
525 Emamectin B1b
526 Phosalone
527 Cyclosulfamuron
528 Bifenox
529 Piperophos
530 Quizalofop-methyl
531 Indoxacarb
532 Flufenoxuron
533 Spinosyn A
534 Pyraclostrobin
535 Triazoxide
536 Amisulbrom
537 Piperonyl-butoxide
538 Nitralin
539 Fenoxaprop-ethyl
540 Fenoxaprop-P-ethyl
541 Fluoroglycofen-ethyl
542 Diclofop-methyl
543 Fluazuron
544 Benzoximate
545 Captafol
546 Emamectin B1a
547 Pyrimidifen
548 Cyflumetofen
549 Dialifos
550 Quinoxyfen
551 Chlorpyrifos
552 Methoprene
553 Lactofen
554 Quizalofop-P-ethyl
555 Quizalofop-ethyl
556 Imibenconazole
557 Spiromesifen
558 Furathiocarb
559 Butralin
560 Cloquintocet-mexyl

129 Pyracarbolid
130 Diuron (DCMU)
131 Fenamiphos-sulfoxide
132 Carbofuran
133 Isoprocarb
134 Carbaryl (NAC)
135 Metobromuron
136 Chlorsulfuron
137 Quinoclamine
138 Buturon
139 Fenamiphos-sulfone
140 Thiacloprid
141 Ancymidol
142 Isoproturon
143 Amidosulfuron
144 Propanil
145 Demeton-S-methyl
146 Thiofanox
147 Desmetryn
148 Clodinafop (free acid)
149 Ethiofencarb
150 Simetryn
151 Malaoxon
152 Fomesafen
153 Thiophanate-methyl
154 Metsulfuron-methyl
155 Rabenzazole
156 Triforine (isomer)
157 Secbumeton
158 Prometon
159 Cycluron
160 Thifensulfuron-methyl
161 Carboxin
162 Florasulam
163 Mephosfolan
164 Lenacil
165 3,4,5-Trimethacarb
166 Propazine
167 Flutriafol (isomer)
168 Sebuthylazine
169 Hexazinone
170 Linuron
171 Fenobucarb
172 Furametpyr
173 Haloxyfop
345 Flufenacet
346 Zoxamide
347 Malathion
348 Flamprop-methyl
349 Oryzalin
350 Tebufenozide
351 Penconazole
352 Bixafen
353 Bifenazate
354 Terbucarb
355 Ametoctradin
356 Orthosulfamuron
357 Fenoxanil
358 Isoxaben
359 Triazamate
360 Metconazole
361 Isoprothiolane
362 Isazofos
363 Triflumuron
364 Tepraloxydim (isomer)
365 Mandipropamid
366 Pyrazosulfuron-ethyl
367 Iprobenfos
368 Benzyldimethyldodecyl-ammonium Chloride
369 Vernolate
370 Pebulate
371 Hydramethylnon
372 Acibenzolar-S-methyl
373 Benazolin-ethyl
374 Flusilazole
375 Cyprodinil
376 Alachlor
377 Etaconazole
378 Isofenphos-methyl
379 Bupirimate
380 Epoxiconazole
381 Metolachlor
382 Furmecyclox
383 Pethoxamid
384 Butylate
385 Tris(2-chloro-1- (chloro-methyl)ethyl)phosphate
386 Fenazox
387 Fenoxycarb
388 Pyrifenox (E, Z)
389 Triazophos
561 Etoxazole
562 Chlorfluazuron
563 Spinosyn D
564 Pyriproxyfen
565 Brodifacoum
566 Ethion
567 Flumetralin
568 Pyrazophos
569 Propargite
570 Fluthiacet-methyl
571 Coumaphos
572 Pyributicarb
573 Acequinocyl
574 Carbophenothion
575 Spinetoram J
576 Hexythiazox
577 Prothiophos
578 Isopropalin
579 Pendimethalin
580 Spirodiclofen
581 Bromophos-methyl
582 Fenpropathrin
583 Acrinathrin
584 Spinetoram L
585 Cyhalothrin (isomer)
586 Propaquizafop
587 Flucycloxuron
588 Flucythrinate
589 Bioresmethrin
590 Resmethrin
591 Rotenone
592 Fenazaquin
593 Tetramethrin
594 Avermectin B1a
595 Cyphenothrin
596 Cypermethrin (isomer)
597 (E)-Fenpyroximate
598 Fluvalinate
599 Phenothrin (isomer)
600 Temephos
601 Pyridaben
602 Bifenthrin
603 Pyridate
604 Permethrin
605 Deltamethrin

174	Siduron	390	Sulfallate	606	Pyridalyl
175	Ethiprole	391	Thiazopyr	607	Profoxydim
176	Oxadixyl	392	Spirotetramat	608	Ivermectin
177	Terbumeton	393	Diazinon	609	Doramectin
178	Benodanil	394	Etrimfos	610	Cinidon-ethyl
179	Desmedipham	395	Napropamide	611	Moxidectin
180	Terbuthylazine	396	Phosmet	612	Etofenprox
181	Naled (Dibrom)	397	Chromafenozide	613	Bentazone
182	Heptenophos	398	Azinphos-methyl	614	Gibberellic acid (Gibberellin)
183	Methabenzthiazuron	399	Ethoxysulfuron	615	4-Chlorophenoxyacetic acid
184	Fosthiazate (isomer)	400	Novaluron	616	2,4-D (2,4-PA)
185	Spiroxamine	401	Fenbuconazole	617	MCPA (MCP)
186	Pirimicarb	402	Pyridaphenthion	618	2-Naphthoxy acetic acid
187	Phorate-sulfoxide	403	Ipconazole	619	Dichlorprop
188	Propyzamide	404	Dimoxystrobin	620	Bromoxynil
189	Chlorbromuron	405	Bensulfuron-methyl	621	2,4,5-T
190	Ametryn	406	Chlorpyrifos-oxon	622	Mecoprop-P
191	Phenmedipham	407	Propisochlor	623	Triclopyr
192	Deet	408	Fluquinconazole	624	Terbacil
193	Methfuroxam	409	Bitertanol (diastereo isomers)	625	Cyclanilide
194	Halofenozide	410	Pyriminobac-methyl (E)	626	Fenoprop
195	Pirimicarb-desmethyl-formamido	411	Cadusafos	627	DNOC
196	Probenazole	412	Sulfotep	628	Ioxynil
197	Paraoxon-ethyl	413	Chlorimuron-ethyl	629	Acifluorfen
198	Paclobutrazol	414	Fonofos	630	Fludioxonil
199	Isonoruron	415	Dimethomorph (E, Z)	631	Fipronil
200	Cinosulfuron	416	Triflusulfuron-methyl	632	Fipronil-sulfide
201	Disulfoton-sulfoxide	417	Tolylfluanid	633	Diclofop
202	Dimefuron	418	Cyazofamid	634	Nicarbazin
203	Nicosulfuron	419	Mecarbam	635	Fipronil-sulfone
204	Azamethiphos	420	Carfentrazone-ethyl	636	Dinoseb
205	Thionazin	421	Propaphos	637	Dinoterb
206	Promecarb	422	Butafenacil	638	Endosulfan-sulfate
207	Norflurazon	423	Isomethiozin	639	Hexaflumuron
208	Propachlor	424	Bensulide	640	Fluazinam
209	Triadimenol (isomer)	425	Flamprop-M-isopropyl	641	Lufenuron
210	Diethofencarb	426	Flamprop-isopropyl	642	Meptyldinocap
211	Tricyclazole	427	Quinalphos	643	Bromadiolone
212	Methacrifos	428	Tetrachlorvinphos (CVMP)	644	Noviflumuron
213	Fenpropidin	429	Chlorfenvinphos (E, Z)	645	Teflubenzuron
214	Prosulfuron	430	Isofenphos	646	Dithianon
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