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Moving mammals

The only personal, tangible experience I have had that relates in any way to the 1997 Kyoto climate treaty (official lifespan: 2002-2012, so now assigned to the dustbin of history…) are the ‘Kyoto cookies’ on the shelves of my local supermarket. I can’t remember whether the point about them was the ‘sustainability’ of their ingredients or that they were non-fattening, or perhaps it was something else – for example an especially tasty biscuit brought back from Kyoto by someone attending a congress there and recreated by a Dutch baker. What was clear, though, was that ‘Kyoto’, despite all the scepticism about the need for and ‘realism’ of treaties like this (even back then), was nonetheless passed down to the proverbial man in the street and imbued with meaning by a supermarket chain – not the most conventional approach to educating the public on nature and the environment.

Over the past few decades the world, and particularly the western world, has been confronted with a series of far-reaching developments, including the ICT revolution, our proven vulnerability to terrorist actions and a series of crises in the financial and economic realm and, partly as a result, large-scale, far-reaching geopolitical changes at the global level. One of the effects of the current economic-financial crises is that economic growth has fallen, in some cases to below zero, while positive growth is often regarded as absolutely essential for continued prosperity. In concrete terms, though, it translates to less fossil fuel being burned, falling sales and consumption of all manner of goods – apart from real estate, also luxury items like multiple foreign holiday trips every year – and an easing of road congestion. And so in a roundabout way the goals of green-minded NGOs, for years deemed utopian and therefore out of the question, are nonetheless being realised. So we should count our blessings!

That the original Kyoto targets have not been secured by a long chalk and that in later preliminary talks on follow-up treaties it proved impossible to even reach agreement on basic targets means, among other things, that the process of global climate change that began to unfold in the 1990s continues unabated and that it is now questionable whether humanity still has time to achieve the kind of drastic change required to as yet turn the tide. Perhaps during the forthcoming negotiations the current position of the US president – no longer guided, as he was, by a desire for re-election when setting and implementing a climate agenda – will contribute to greater, and more structural, success when it comes to formulating and then actually securing climate targets. And perhaps now that the BRIC nations (Brazil, Russia, India and China) are also becoming more wealthy these countries will also show rather more responsibility in this arena and translate that responsibility into concrete, practical action. It would certainly be worth the effort.

In the context of policies to mitigate climate change and limit its consequences there exists something of a conundrum in the Netherlands, too. When 'new' species move into the Netherlands as a result of climate change, this is regarded as a more or less natural process and thus one that does not necessarily need to be steered or controlled. Species arriving in the Netherlands with human aid (whether intentionally or unintentionally) are seen as intruders, on the other hand. If there is any risk of them 'endangering' the country's present biodiversity, its public health or economic values, they are even regarded as a threat – in which case the species in question must be exterminated. But what's the difference between mankind's role in species introduction due to climate change and species introduction via the bilge water of a mammoth tanker? Both may be unintended, but in both cases it is *Homo sapiens* that is the ultimate vector.

Why this distinction and, while we're on the subject, why that sometimes so panicky response to the emergence or discovery of another newcomer? The appearance of 'new' species in the Netherlands has always been with us, the brown rat (*Rattus norvegicus*) being just one example of a mammal species within historical times. This is generally happened or happens unnoticed, without any problems at all, let alone that a need was felt to take combative action. In the early days there may be some teething troubles, as new species often become established according to a fixed pattern: first a pronounced and rapid increase in the number of individuals, then generally a gradual decline and eventually stabilisation at a level acceptable in every respect. No worries, then! But this comforting knowledge, this empirical evidence, is not usually employed in any rational way when it comes to programmes addressing new species. On the contrary, the situation is presented as if doomsday lies just round the corner unless immediate and radical action is taken.

Even with naturally occurring arrivals there's still plenty of moaning and groaning, though not always from the same quarters. Once more, the short-term perspective is dominated by visions of doom. Thus the wolf (*Canis lupus*) and the lynx (*Lynx lynx*) are not welcome at all in certain circles, even though it's widely accepted that precisely these species, as top predators, probably mean a welcome addition to present-day biodiversity in the Netherlands and will cause very little damage, most of it merely temporary or incidental.

As is the case with strategies for species protection, with exotic and other new species, too, what should be the prime area of focus are numbers and population trends. The next step is to identify the factors driving those trends. Where in the case of threatened species the main focus is generally on the negative consequences of human activity for the species in question, with exotic species the focus should then be on the risks for 1. nature, in particular biodiversity (the ecological risks), 2. material damage to real estate and other economic goods (the economic risks), and 3. public health (the humanitarian risks).

And while we're on the subject: one overtly active and much-coveted form of mammal movement by us humans is species reintroduction. Here, government agencies and other parties must guarantee that the causes of local extinction have been removed and that supporting measures to that end have been duly implemented. With reintroduction of the otter (*Lutra lutra*) in the Netherlands, that proved not to be the case. Despite the relatively large number of animals ending up as roadkill, though, the reintroduction programme can nonetheless be termed a success. More controversial, perhaps, is all the carting-around of animals captured elsewhere, the frequent use of sedatives, and the use of transmitters and other devices attached to or implanted in the creatures' bodies.

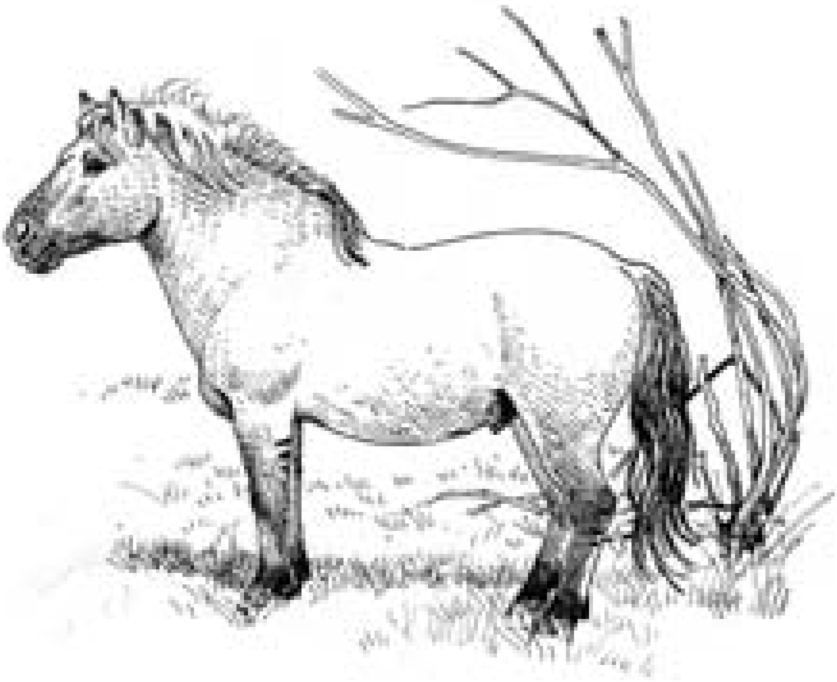
The kind of insight yielded by carefully conducted research on an 'exotic' species is demonstrated by Mulder's case study on Dutch occurrence of the raccoon dog (*Nyctereutes procyonoides*), published in this edition of *Lutra*. The basic procedure is straightforward: first, track down potential problems as carefully as possible; next, based on the facts, examine what, if any, negative consequences are to be anticipated; then, where applicable, identify the potential for intervention; and, finally, take a decision on the basis of a cost-benefit analysis. 'Control' then generally emerges as one of the least attractive options, and this is the case with the raccoon dog, too. Indeed, control is often impossible in practice (lack of physical means, unfeasibly large-scale and thus too expensive or with too many side-effects) and, more importantly, alternative strategies are often cheaper and more effective, such as compensation for damage or intensification of management activities directed towards the species in question. In this context it's of interest to note that a long-term study was started this year on the muskrat (*Ondatra zibethicus*), an exotic species now common throughout most of the Netherlands. The species is to be monitored in 177 5x5 km grid squares, one-third of these with 'pest controllers' working at traditional intensity, one-third with efforts stepped up and one-third with efforts ratcheted down. How does the muskrat population develop in each case? How much damage is there to dikes and the banks of watercourses? What are the costs of repair? These are some of the questions to be answered in this study, which will probably vastly improve our knowledge of this exotic species and provide practical handles for future policy.

The movement of species and the appearance of 'new' species in a particular country are part and parcel of nature, and therefore also of mammalian life, and are consequently an important issue for scientific researchers. Species are, in principle, always on the move, looking for a more favourable niche in terms of climate, food, cover or whatever. Besides the raccoon dog, this proves to hold true for all the other species featured in the articles in this issue of *Lutra*: the harbour porpoises (*Phocoena phocoena*) coming to explore the Marsdiep tide-race, the colonisation of new polders by the common vole (*Microtus arvalis*) and the temporal changes in the geographic distribution of porpoises on the continental shelf.

Moving mammals – open to three interpretations, all of them covered or reflected in this issue!

Meanwhile, Kees Camphuijsen has stepped down from *Lutra*'s editorial board. For over six years Kees has made a highly valued contribution to the journal, not only through his lucid, succinct and always closely argued commentaries and the professional, accurate and stimulating support he gave to authors, but also as an author himself. In the period of his editorship Kees was (co-)author – and generally first author – of six full papers and one short note. In earlier years Kees had already been a regular contributor to *Lutra* and he has stated his intention to remain faithful. We would like to take this opportunity to thank Kees once more for all he has contributed.

Kees J. Canters



The long-term influence of grazing by livestock on common vole and raptors in man-made wetlands in the Netherlands

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Abstract: Several studies have examined the effects of grazing by wild ungulates or livestock on the abundance of small mammals; some studies have also examined the effects on the abundance of raptors feeding on small mammals. In most studies the abundance of small mammals was negatively affected by grazing, while raptors were found to show a numerical response to the density of small mammals. However, most studies rely on census data from time spans of just 1-4 years. Because there are often large fluctuations in the numbers of small mammals, there is a need for long-term studies. In this study we analyse the long-term effects (3-27 years) of grazing by livestock on vegetation development, vegetation structure, common vole (*Microtus arvalis*) index and density of vole-feeding raptors in man-made wetlands in the Netherlands. The man-made wetlands studied are characterised by a low level of physical perturbation, and without additional management measures their vegetation, of short grasses, will soon be replaced by tall vegetation dominated by reed (*Phragmites australis*), wood small-reed (*Calamagrostis epigejos*) and shrubs. Grazing was initiated while short grasses dominated the vegetation. The intense grazing (summer grazing with a stocking rate of more than 0.8 animals.ha^{-1}), created a homogeneous short vegetation. Grazing with a low stocking rate (year-round grazing with a stocking rate less than 0.6 animals.ha^{-1} or summer grazing with a stocking rate of less than 0.1 animals.ha^{-1}) led to a heterogeneous vegetation. However, after a few years, relatively sharp boundaries developed between short-grazed grassland and closed reed stands and the intermediate stage, characterised by a moderate reed height (between ca. 0.5-1.5 m), increasingly disappeared. In grazed areas relatively high densities of common voles (vole indices 15-35 voles / 100 trap nights) were restricted to parts with a moderate reed height. Such areas only temporarily existed and their disappearance led to a decrease of vole abundance after some years. Vole-feeding raptors showed a numerical response to changing vole densities. Vegetation structure also had an effect on raptor density. Maximum raptor densities were found at sub-maximum vole indices, where average reed height was somewhat lower. The relevance of grazing as a tool for management of vole-feeding raptors in man-made wetlands is highly dependent on the potential of grazing to revert tall vegetation to an earlier successional stage. The regular occurrence of high vole densities and their predators may be achieved by creating a cyclic variation in stocking rates. Years with relatively low stocking rates should be alternated with some years with higher livestock densities.

Keywords: livestock grazing, long-term effects, common vole, *Microtus arvalis*, vole-feeding raptors, reed height, wetlands.

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Introduction

As a result of large hydraulic-engineering works intended to protect against flooding or to reclaim land for agriculture, about 30,000 ha of new wetlands have been established in the Netherlands during the last century (Schultz 1992). Many of these man-made wetlands are of international conservation importance because of the occurrence of large numbers of birds. Many of these bird species, including vole-feeding raptors and owls, are attracted by early stages in vegetation succession (van Eerden 1984, Dijkstra et al. 1995). Since the level of perturbation, such as inundation by salt or freshwater, erosion by ice, or grazing by ungulates, is very low in these wetlands, these early successional stages only exist for a short while. Livestock grazing is one of the management options for stopping or slowing down vegetation succession (e.g. Bakker 1989, Scherfose 1993), with the aim of maintaining early successional stages and their characteristic plant and animal species.

The effects of grazing on vegetation structure and habitat use by birds have attracted a great deal of attention (e.g. Larsson 1969, Soikkeli & Salo 1979, Holechek et al. 1982, van Wieren 1991, Duncan 1992, Vulink & van Eerden 1998). The effects of grazing by wild ungulates (Keesing 1998, Smit et al. 2001) or livestock (Grant et al. 1982, Bock et al. 1984, Heske & Campbell 1991, Hayward et al. 1997, Schmidt et al. 2005, Wheeler 2008, Johnson & Horn 2008, Bakker et al. 2009) on the abundance of small mammals and their avian predators have been less well studied. In most studies the abundance of small mammals was found to be negatively affected by grazing, while raptors were found to show a numerical response to small mammal density. However, except for the work of Hayward et al. (1997) and Bakker et al. (2009), these studies rely on census data from just 1-4 years. Because small mammals often show large fluctuations in numbers, there is a need for long-term studies (Hayward et al. 1997). Microtine rodents are known to show large variation in population size; often these fluctuations are cyclic with peaks every 3-4 years (Krebs & Myers 1974, Hansson & Henttonen 1985). In the Netherlands populations of common voles (*Microtus arvalis*), the main prey species for the majority of raptor species (Dijkstra et al. 1995), are weakly cyclic (van Wijngaarden 1957, Cavé 1968, Dijkstra & Zijlstra 1997). The numerical responses of vole-feeding raptors to changing vole densities may occur either rapidly, without an obvious time lag (Korpimäki & Norrdahl 1989, Korpimäki & Norrdahl 1991, Korpimäki 1994) or with a long delay (Keith et al. 1977, Erlinge et al. 1983).

We examined the long-term effects of grazing by cattle and horses on the abundance of common vole, vole-feeding raptors and one species of owl (hereafter referred to as raptors) in a long-term study (27 years). The effect of vegetation development on the density of raptors is analysed for the entire study period (1969-1995). The effects of (1) grazing on vegetation structure, (2) vegetation structure on vole numbers, and (3) vegetation structure and vole numbers on raptor abundance, are analysed for the second part of the study period (1983-1995).

Methods

Study areas

The study was conducted in two recently reclaimed areas in the Netherlands: Lauwersmeer (53°20'N, 6°10'E) and Oostvaardersplassen (52°26'N, 5°19'E) (figure 1). The Lauwersmeer polder (9100 ha) was reclaimed from the Wadden Sea in 1969. The nature reserve (4500 ha) consists of former tidal flats (2100 ha; hereafter referred to as flats), former accretion works (300 ha) and shallow and deeper waters (2100 ha). Soil types, varying from loamy sand to clay-rich, are related to the elevation of the flats: the higher-lying flats being more sandy than the less-elevated ones. After empoldering, the soils of the flats gradually

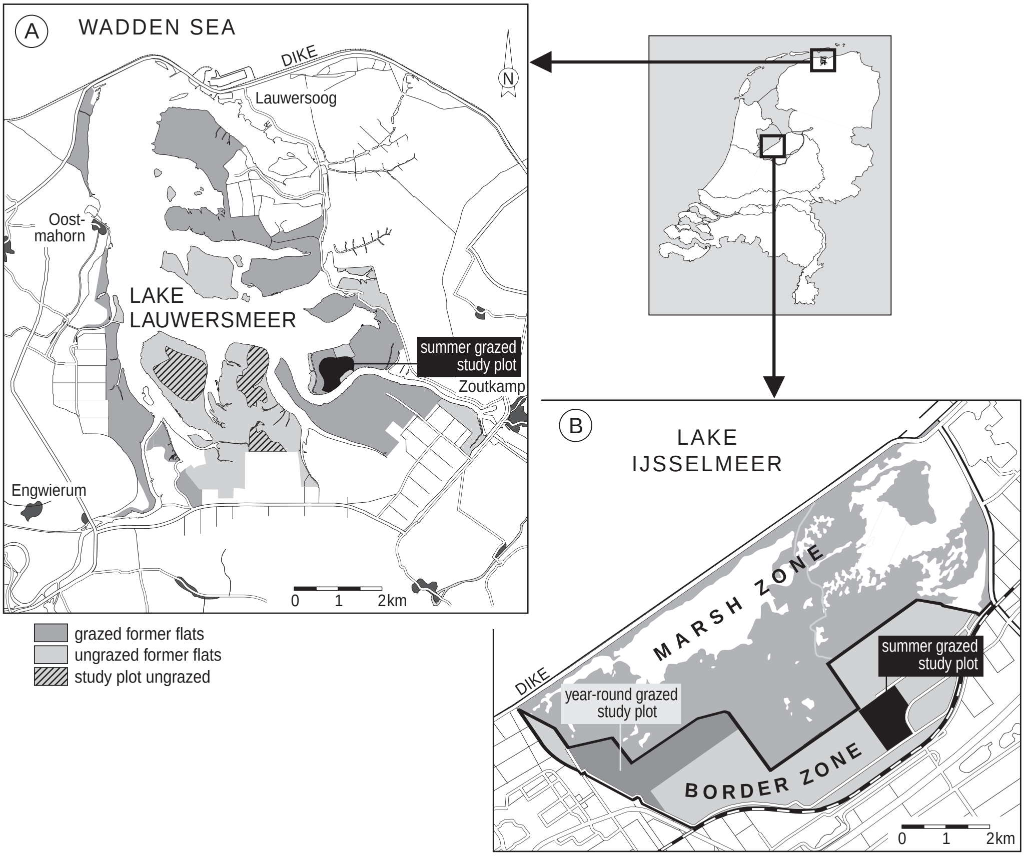

Figure 1. Overview of the study areas in the Netherlands: (A) Lauwersmeer (a former estuary) and (B) Oostvaardersplassen (along lake IJsselmeer, the former Zuiderzee).

desalinated (Joenje 1978, van Rooij & Drost 1996). In the first few years after empoldering, an undisturbed vegetation succession took place (Joenje 1978). In 1982, summer grazing with cattle and horses was started on parts of the flats (1300 ha).

The Oostvaardersplassen study area (5600 ha) is located in southern Flevoland, a polder reclaimed in 1968 from the freshwater lake IJsselmeer. The nature reserve consists of a central reed marsh (3600 ha) and a well-drained border zone (2000 ha). Soils in the Oostvaardersplassen are clayey and therefore more fertile than in Lauwersmeer. In the border zone, grasslands (900 ha) were created on former arable fields by sowing grass mixtures (for a detailed description see Vulink & van Eerden 1998). Some areas in the border zone were managed by summer grazing with cattle and horses between 1982 and 1993. From 1984 onwards, year-round grazing with Heck cattle (*Bos taurus*, crossbred from primitive races) and konik horses (*Equus ferus*, a primitive breed of horse originating from Poland) has taken place. The area with year-round grazing gradually expanded in accordance with growth of the herds.

Study plots

In Lauwersmeer, common voles are more numerous on high-lying flats (>0.65 m above target level) than on less-elevated flats (<0.65 m

Table 1. Study plot characteristics of the summer-grazed and ungrazed study plots in Lauwersmeer, and the summer-grazed and year-round grazed study plots in Oostvaardersplassen.

	Lauwersmeer		Oostvaardersplassen	
Grazing regime	Summer grazing	Ungrazed	Summer grazing	Year-round grazing
Study period	1983-1995	1989-1993	1991-1993	1989-1995
Size (ha)	33	190	120	288
Stocking rate (animal.ha^{-1})	0-1.1	0	1.4-1.7	0.3-0.9
Annual grazing period	1982-1992; 7 June to 30 Sept. 1993-1995; 1 May to 31 Oct.	n.a.	1 May to 31 Oct.	Year-round

above target level). For example, in October 1983 the density of burrows was 65.ha^{-1} on high-lying flats versus 2 burrows.ha^{-1} on less-elevated flats. The reason for this is that low-lying flats are regularly flooded for several days, whilst only parts of the high-lying flats are occasionally inundated. The study on the relationship between grazing, vegetation structure, vole density and abundance of raptors was carried out on high-lying flats of the nature reserve, some grazed and others ungrazed (figure 1A). For the most important characteristics of the study plots, see table 1.

In Oostvaardersplassen, grasses were sown in the summer-grazed study plot in 1989. The vegetation was mown for a period of two years (twice a year), after which the area was summer grazed at high stocking rates for three years (for characteristics of the study plot, see table 1). In the year-round grazed study plot in Oostvaardersplassen grasses were sown in 1982. In the period 1984 to 1988 the density of Heck cattle and konik horses was relatively low and grassland was also mown; 1989 was the first year of grazing without additional mowing (for characteristics of the study plot, see table 1). The locations of the summer-grazed and year-round grazed study plots in Oostvaardersplassen are shown in figure 1B.

Vegetation development

Long-term vegetation development on flats in Lauwersmeer was derived from vegetation maps (scale 1:5000 or 1:10,000) based on the interpretation of satellite photographs (1972, 1975) and aerial photographs (1980, 1984 and 1989), combined with ground surveys that identified the different zones by using data from quadrates (Küchler & Zonneveld 1988). The vegetation composition in 1995 was based on field visits. The original vegetation maps distinguished between 10 vegetation types. In this study, vegetation types are grouped into three categories: vegetation of halophytic pioneers, dominated by glasswort (*Salicornia* spp.), with herbaceous seepweed (*Suaeda maritime*), lesser sea-spurrey (*Spergularia marina*) and greater sea-spurry (*Spergularia media*); vegetation of short grasses, dominated by creeping bentgrass (*Agrostis stolonifera*) and common saltmarsh-grass (*Puccinellia maritima*), with red fescue (*Festuca rubra*), marsh foxtail (*Alopecurus geniculatus*) and reed; tall vegetation, dominated by reed, with wood small-reed (*Calamagrostis epigejos*) and willows (*Salix* spp.). The scientific nomenclature of the plant species follows van der Meijden (1996).

As succession proceeds vegetation dominated by short grasses gradually changes into tall vegetation, dominated by reed on desalinated former flats in Lauwersmeer as well as on grassland in Oostvaardersplassen. Since there is a significant correlation between reed cover and reed height (Huijser et al. 1996), the latter was used as an index for the vegetation structure in the study plots. Reed height was measured in autumn, using a measuring rod. At each vole-trapping station, five measurements were taken with a total of 100 or 150 readings per study plot. Before 1988, measurements in Lauwersmeer were based on visual estimations (estimated to the nearest 5 cm).

Developments in the vegetation structure of short grasses in Oostvaardersplassen in 1989-93 were described in terms of average sward height. Sward height was measured by using a polystyrene disc (radius 50 cm, weight 320 gram). The disc was gently lowered on to the sward, and the height of the vegetation was read off on the measuring staff in the centre. At each vole trapping station, the height was measured five times. The 100 to 150 readings per study plot were averaged.

Vole densities

Vole densities were measured two or three times a year (March, July and October). Small mammals were caught according to the modified method of Hörnfeldt (1978). In each study plot, trap lines were set up at 30 m intervals and randomly assigned to each of the three trapping periods per year. On each trap line 10 trapping stations were situated at 10 m intervals, with five traps at each trapping station. Traps were controlled once a day, for a period of three days. Consequently, the number of trap nights was 150 per trap line. Usually, two trap lines were set up per study plot. In the summer-grazed study plot in Lauwersmeer, the number of trap lines was three after 1985, in the year-round study plot in Oostvaardersplassen the number of trap lines was four in all years. In all study plots, common voles made up more than 95% of the small mammals caught. The vole index was defined as the number of common voles caught per 100 trap nights.

Raptor densities

Counts of raptors in Lauwersmeer were initiated in 1969, the year of reclamation; counts in Oostvaardersplassen were started in 1982, fifteen years after empoldering was completed. In both study areas, counts were organised about once a month. In Lauwersmeer the raptor species (partly) feeding on common voles are hen harrier (*Circus cyaneus*), buzzard (*Buteo buteo*), rough-legged buzzard (*Buteo lagopus*), kestrel (*Falco tinnunculus*) and short-eared owl (*Asio flammeus*). Hen harrier and rough-legged buzzard are mainly present in winter, while buzzard, kestrel and short-eared owl are present throughout the year. Data of the two most common species (hen harrier and kestrel) were selected for detailed analysis in the study plots. In the study areas, the diets of these two species consisted of more than 90% of common voles (Masman et al. 1988, Dijkstra et al. 1995).

Raptor densities were expressed as bird-days.ha^{-1} per year. The number of bird-days was calculated for each interval between consecutive counts as the average of these two counts multiplied by the interval length in days. These values were added up for each year (months from July till June). For the entire flats in Lauwersmeer all birds were included, whilst in the study plots only flight-hunting birds were selected.

Results

The effects of grazing on vegetation structure

On Lauwersmeer flats with a natural succession of the vegetation, parallel to the process of desalination, halophytic pioneers were gradually replaced by short grasses, which in their turn were replaced by tall vegetation (figure 2A). Flats where summer grazing was initiated were, on average, in an earlier successional stage at the onset of grazing than flats which remained ungrazed (cf. figures 2A and 2B). After the initiation of grazing, the successive decrease in the incidence of short grasses and of halophytic pioneers halted for some years, this was in contrast to a steady

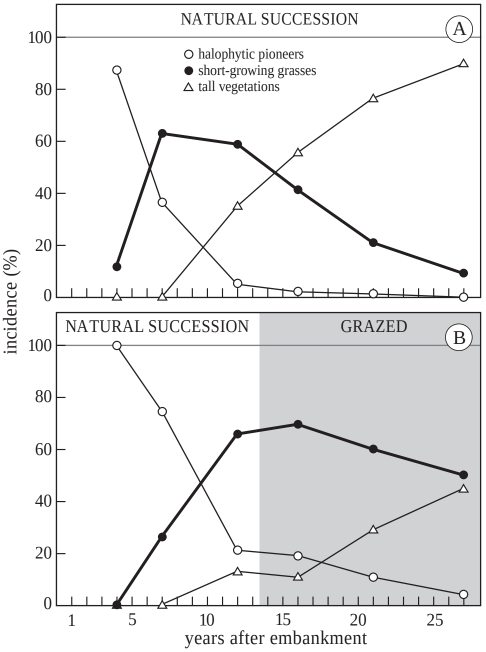

Figure 2. Vegetation succession on former tidal flats in the Lauwersmeer nature reserve (year 1 = 1969): (A) flats with a natural succession of the vegetation (1030 ha), (B) flats with summer grazing (since year 14; 890 ha). The percentage of each vegetation type is presented (= % incidence) for years 4, 7, 12, 16, 21 and 27 after empoldering.

decrease on the ungrazed flats. Thereafter, the proportion of halophytic pioneers and short grasses decreased again, while the proportion of tall vegetation increased. However, even in the last year of the study period, the proportion of tall vegetation was much lower than on ungrazed flats (figures 2A and 2B).

In the summer-grazed study plot in Lauwersmeer, grazing transformed a vegetation of mosaics of reed and short grasses into a large-scale open vegetation. In the first five years of grazing, when the average stocking rate was relatively high (figure 3A1), the average reed height was lower than 0.5 m (figure 3B1). After the fifth year, when the stocking rate was very low, the average reed height sharply increased and gradually levelled off. In the last years of the study period, reed height was only slightly lower than on comparable flats in the ungrazed study plot at Lauwersmeer (1.4 m versus 1.6 m). Gradually, the vegetation in the summer-grazed study plot changed into a closed reed stand, comparable to the ungrazed study plot. On less-elevated flats the vegetation remained open: the average reed height was lower than 0.7 m for the entire period of grazing.

In the summer-grazed study plot in Oostvaardersplassen, grazing at high stocking rates (cf. table 1) resulted in a homogeneous short grassland with a very low reed height (less than 0.1 m) for the entire study period. Average sward height showed only a slight increase in summer (table 2A).

In the year-round grazed study plot in Oostvaardersplassen the average stocking rate strongly increased from about 0.3 to 0.9 animals.ha^{-1} during the study period (table 1). The average stocking rate increased from 0.3 to 1.1 animals.ha^{-1} in summer, but was rather stable in winter (in most years 0.3-0.6 animals.ha^{-1}; figure 3A2). Cattle and horses did not distribute equally over the study plot. In a part of the study plot the animal density was extremely low from the first year of grazing without additional mowing. The area with a reed height higher than 1.5 m therefore increased from 0% in year 1 to 30% in year 7. The gradual increase in stocking rate during the study period resulted in an expansion of heavily grazed grassland (reed height 0-0.5 m) from 40% in year 1 to 60% in year 7. In the same period the percentage of moderately grazed grassland (reed height 0.5-1.5 m) decreased from 60% to 10%. In the last two years of the study period, the area mainly consisted of intensively-grazed grassland (60%) and ungrazed closed reed stands (30%) with rather sharp boundaries between the two. Maximum reed height in the reed stands in Oostvaardersplassen was higher than in Lauwersmeer, due to higher soil fertility. Along the vole-trap lines, situated in the part of the study plot with a moderate grazing intensity, average reed height increased from 0.6 m in the first year of grazing to 0.8-1.0 m

Table 2. A. Average sward height (cm, ±sd) in March, July and October in the summer-grazed and a part of the year-round grazed study plot in Oostvaardersplassen. Measurements were made at vole-trap lines. B. Average vole index (±sd) in March, July and October in the summer-grazed and a part of the year-round grazed study plot in Oostvaardersplassen (*n* refers to the number of trap lines).

A	Sward height	
	Summer grazing	Year-round grazing
March	6.7±7.2 (1992-93; *n*=100)	5.1±8.4 (1991-93; *n*=150)
July	18.8±15.5 (1991-92; *n*=100)	38.6±17.7 (1990-92; *n*=150)
October	6.5±7.4 (1991-93; *n*=150)	19.5±14.1 (1989-93; *n*=250)

B	Vole index	
	Summer grazing	Year-round grazing
March	0.0±0.0 (1992-93; *n*=4)	0.2±0.3 (1992-93; *n*=4)
July	1.0±1.2 (1991-92; *n*=4)	2.5±2.9 (1991-92; *n*=4)
October	0.7±0.6 (1991-93; *n*=6)	12.4±9.7 (1991-93; *n*=6)

in years 3-4, and decreased to about 0.3-0.4 m in years 5-7 (figure 3B2).

There was more seasonal variation in sward height in the year-round grazed study plot than in the summer-grazed study plot (table 2A). In the year-round grazed study plot cattle and horses minimised sward height outside the closed reed stands during the winter, which led to short grassland in early spring. In March, the average sward height was significantly lower than in the summer-grazed study plot (Mann-Whitney U test, *P*<0.001). During summer, sward height in the year-round grazed study plot increased sharply and the average sward height in July and October was significantly higher than in the summer-grazed study plot (Mann-Whitney U test, *P*<0.001 for both periods).

The effect of vegetation structure on vole density

In the summer-grazed study plot in Lauwersmeer, in the first years of grazing, when the stocking rate was relatively high and vegetation was short, the vole index was relatively low (years 2-5: average vole index 2.9±1.5; *n*=9 trap lines; figure 3C1). The vole index increased after the stocking rate was lowered and the reed height increased, and remained high for five consecutive years (years 6-10: average vole index 15.1±7.5; *n*=15 trap lines). In later years, when the reeds were high, the vole index decreased again to a relatively low level (years 11-14: average vole index 3.6±4.6; *n*=12 trap lines). On comparable flats in the ungrazed study plot, the vole index was very low (years 9-12: average vole index 0.3±0.7; *n*=4 trap lines). Considered over the entire study period of fourteen years, high vole indices were restricted to areas with a moderate reed height (between ca. 0.5-1.5 m; figure 4A). The variation in vole density in the summer grazed study plot did not follow the multi-annual cycle found in the border zone of Lauwersmeer Nature Reserve (figure 3C1).

In Oostvaardersplassen the vole index in the year-round grazed study plot showed a larger seasonal variation than in the summer-grazed study plot (table 2B). In the year-round grazed study plot, the vole index increased significantly from March to October (ANOVA: $F_{1,12}$=8.02, *P*<0.05). In the summer-grazed study plot, the vole index was low in all months of trapping and the increase from March to October was not significant. The difference in vole index between the two study plots was only significant for October (Mann-Whitney U test, *P*<0.01). In the year-round

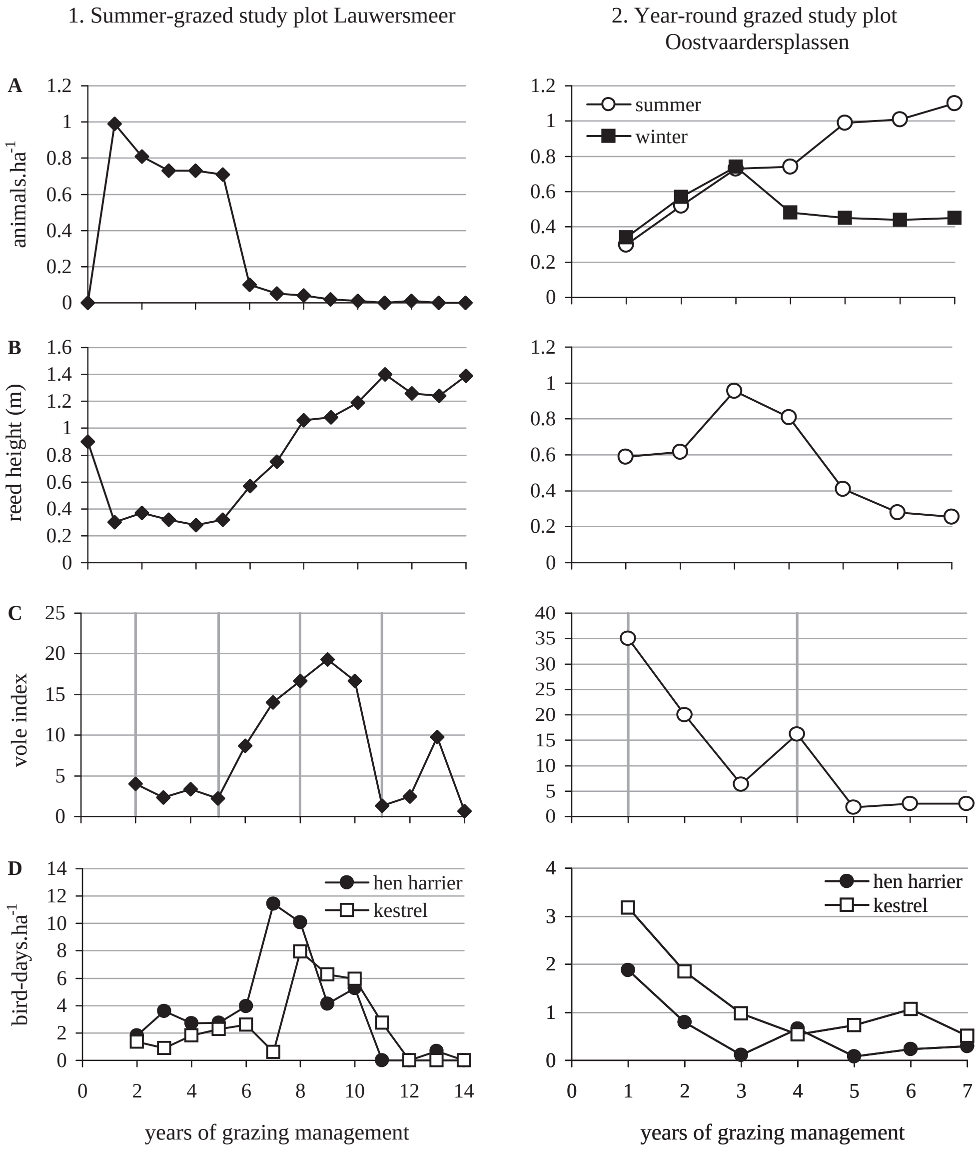

Figure 3. Response of vegetation structure to grazing and consequences for the vole index and raptor density in (1) the summer-grazed study plot in Lauwersmeer (33 ha; grazing by grazing) and (2) the year-round grazed study plot in Oostvaardersplassen (288 ha; grazing by cattle and horses). Data are all expressed as a function of years of grazing management (Lauwersmeer: year 1 = 1982; Oostvaardersplassen year 1 = 1989). (A) Average stocking rate (in year-round grazed study plot in Oostvaardersplassen shown for summer (April-September) and winter (October-March)), (B) Average reed height in October, (C) vole index in October, vertical bars refer to vole peak-years (based on data for October from the well-drained border zone in the Lauwersmeer nature reserve (Dijkstra et al. 1995) and from the borderzone in Oostvaardersplassen (N. Beemster, unpublished data)), and (D) hen harrier and kestrel density, expressed as bird-days.ha^{-1} per year (1 July - 30 June). Only hunting birds were included.

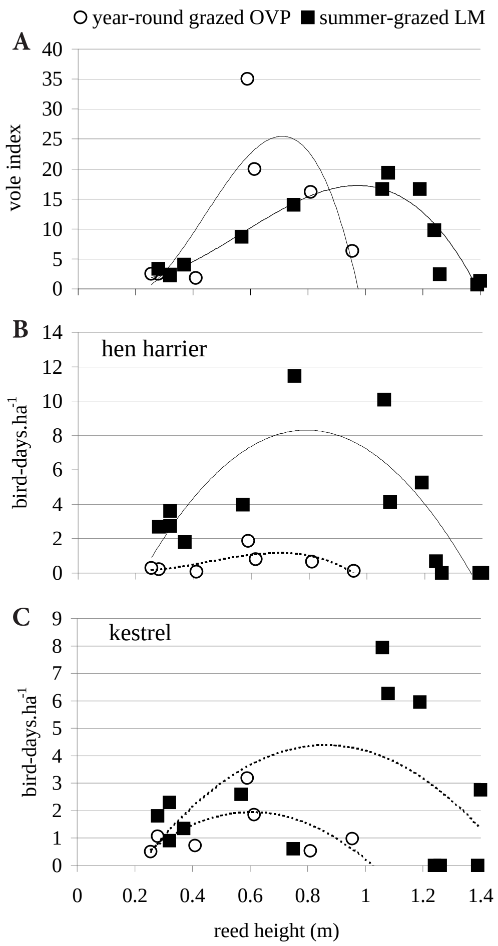

Figure 4. Relation of vole index (A) and raptor density (hen harrier (B), kestrel (C)) to reed height in the summer-grazed study plot in Lauwersmeer (33 ha) and the year-round grazed study plot in Oostvaardersplassen (288 ha). Data are expressed as a function of years of grazing management (summer grazed study plot in Lauwersmeer: year 1 = 1982; year-round grazed study plot in Oostvaardersplassen year 1 = 1989). (A) Vole index in October (summer-grazed study plot in Lauwersmeer: R^2=0.85, P<0.001; year-round grazed study plot in Oostvaardersplassen: R^2=0.69, P<0.05), (B) hen harrier density, expressed as bird-days.ha^{-1} per year (1 July - 30 June). Only hunting birds were selected (summer-grazed study plot in Lauwersmeer: R^2=0.64, P<0.001; year-round grazed study plot in Oostvaardersplassen: R^2=0.54, NS) and (C) kestrel density, expressed as bird-days.ha^{-1} per year (1 July - 30 June). Only hunting birds were selected (summer-grazed study plot in Lauwersmeer: R^2=0.40, NS; year-round grazed study plot in Oostvaardersplassen: R^2=0.21, NS). Insignificant lines are stippled.

grazed study plot, the vole index decreased sharply during the study period (figure 3C2; ANOVA: $F_{1,12}$=23.37, P<0.001). The vole index in the year-round grazed study plot seemed to be somewhat higher in vole peak-years in the borderzone of Oostvaardersplassen in comparison to other years (figure 3C2), however, the differences are not significant because of the small sample sizes. In the summer-grazed study plot, there was no clear change during the three years of study. As with the summer-grazed study plot in Lauwersmeer, high vole indices in the year-round study plot in Oostvaardersplassen were restricted to a moderate reed height (figure 4A). The maximum vole indices in Oostvaardersplassen were higher than in Lauwersmeer.

The effect of vegetation structure and vole density on raptor density

Before grazing was introduced in Lauwersmeer, the numbers of raptors on the flats that were ungrazed throughout, and on flats which were later grazed, were quite similar (figure 5) with the exception of the hen harrier, which showed differences in density over the period 3-13 years after empoldering. These were significantly higher on consistently ungrazed flats (Wilcoxon signed rank test: n=11, P<0.05).

On ungrazed flats with a continuing natural succession of the vegetation, raptor numbers showed a strong increase in the first years

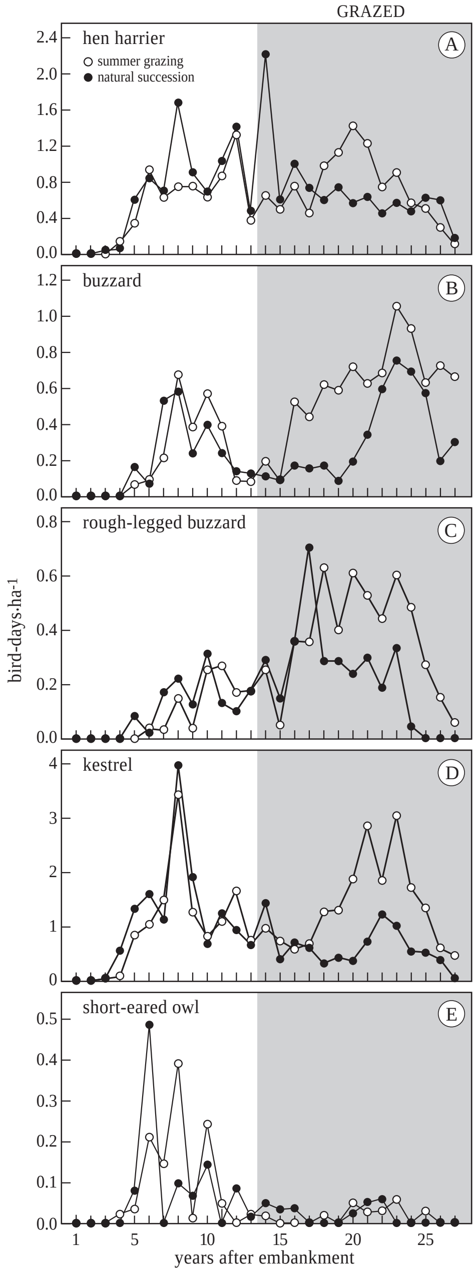

Figure 5. Densities of raptors, depending for a major part of their diet on common voles, on former tidal flats in Lauwersmeer after empoldering (year 1 = 1969); closed dots, flats with a natural succession of the vegetation (1030 ha), open dots, flats with summer grazing (since year 14; 890 ha). (A) hen harrier, (B) buzzard, (C) rough-legged buzzard, (D) kestrel, (E) short-eared owl.

after empoldering, followed by a stabilisation. In the latter half of the study period raptor numbers gradually declined, with some species declining somewhat earlier than others (figure 5). Long-term developments in raptor abundance are related to a change in the extent of the area covered with short grasses (figure 2A) and consequently to a change in food supply of common voles (Dijkstra et al. 1995). Buzzard numbers increased again more than twenty years after empoldering, when a small breeding population became established. These birds mainly find their food outside the ungrazed flats.

At the time grazing was introduced, some species were still numerous (hen harrier and rough-legged buzzard), while others were on the decline (buzzard and kestrel) or had almost disappeared (short-eared owl). After the introduction of grazing, four out of five species showed a temporary increase in number, contrasting with the general decline on the ungrazed flats. The short-eared owl did not respond to this change: the species had almost disappeared before the onset of grazing, and this measure did not result in the return of the species (figure 5E). The following remarks are confined to the four species that responded to the introduction of grazing.

For the entire period of grazing, three out of four species were significantly more abundant on grazed flats than on the consistently ungrazed flats (Wilcoxon signed rank test: n=14 years, buzzard P<0.01, rough-legged buzzard P<0.05 and kestrel P<0.01). However, densities showed a large year-to-year variation. Roughly, three periods can be distinguished. In the first four years of grazing, some species were more abundant on ungrazed flats, while other species were more abundant on grazed flats (the differences are, however, not significant because of the small sample sizes). In the period 5-10 years after the initiation of grazing, raptor densities on grazed flats were relatively high. All the species were now significantly more abundant on grazed flats (Wilcoxon signed rank test: n=6 years, $P<0.05$ for all species). In the last four years of the study period, raptor densities on grazed flats decreased sharply, although the abundance of most species was still higher than on ungrazed flats (but because of the small sample sizes, the differences are not significant).

In the summer-grazed study plot in Lauwersmeer, densities of hen harrier and kestrel were relatively low in the first years after initiating grazing (figure 3D1). Both species were numerous shortly after the stocking rate had been lowered (years 6-10). Later (years 11-14), the densities were again low. Annual variations in the density of hen harrier and kestrel were tested by multiple regression, with average vole index, average reed height + average reed height^2 as independent variables. Variation in raptor density was significantly correlated with vole index (hen harrier: $R^2=0.49$, $P<0.01$; $n=13$ years; kestrel: $R^2=0.51$, $P<0.01$; $n=13$ years) and reed height + reed height^2 (hen harrier: $R^2=0.64$, $P<0.01$; $n=13$ years; kestrel: $R^2=0.62$, $P<0.05$; $n=13$ years). The results for the full model were: hen harrier: $R^2=0.67$, $P<0.05$; $n=13$ years; kestrel: $R^2=0.62$, $P<0.05$; $n=13$ years. Hence, both species of raptors showed a clear numerical response to changing vole densities, although maximum raptor densities were found at sub-maximum vole indices, where average reed height was somewhat lower (cf. figures 4B and 4C with figure 4A).

In the ungrazed study plot in Lauwersmeer densities of hen harrier and kestrel were low for the entire study period (four years). The average number of bird-days hunting.ha^{-1} per winter was 0.23 (range 0.14-0.32) for the hen harrier and 0.50 (range 0.14-0.93) for the kestrel.

In the summer-grazed study plot in Oostvaardersplassen, densities of hen harrier and kestrel were also low for the entire study period (three years). The average number of bird-days hunting.ha^{-1} per winter was 0.36 (range 0-0.85) for the hen harrier and 0.59 (range 0.32-0.76) for the kestrel.

In the year-round grazed study plot in Oostvaardersplassen both species were relatively numerous in the first two years of grazing without additional mowing; in later years densities were much lower (figure 3D2). Both species showed a decrease in density during the study period. Annual variations in the density of hen harrier and kestrel were tested by multiple regression, with average vole index and average reed height + average reed height^2 as independent variables. Variation in raptor density was significantly correlated with vole index (hen harrier: $R^2=0.93$, $P<0.001$; $n=7$ years; kestrel: $R^2=0.75$, $P<0.01$; $n=7$ years). Reed height + reed height^2 did not have an additional effect. Hence, as was the case in the summer grazed study plot in Lauwersmeer, both species of raptors showed a clear numerical response to changing vole densities. However, in contrast to the summer grazed study plot in Lauwersmeer, no effect of reed height + reed height^2 could be detected, probably because of the small sample sizes (number of years). In the summer grazed study plot in Lauwersmeer maximum raptor densities were much higher than in the year-round grazed study plot in Oostvaardersplassen, despite lower maximum vole indices.

Discussion

Effects of grazing on vegetation structure

Intense grazing by livestock of reed stands led to a homogeneous short vegetation, as was also found by Vulink et al. (2000). When, after a period of intense grazing or mowing, grazing with a low stocking rate was practised, patches were grazed with a different intensity and a heterogeneous vegetation gradually developed. The development of a patchy vegetation at a low stocking rate has been documented earlier by Bakker et al. (1984) and Bakker (1989). However, after some years, relatively sharp boundaries between short-grazed vegetation and closed reed stands developed and there were almost no areas with a moderate reed height. Annual variation in vegetation structure was more pronounced in the year-round grazed study plot than in the summer-grazed study plots.

Effects of grazing on vole density

In grazed areas, relatively high densities of common voles were restricted to parts with a specific vegetation structure, characterised by a moderate reed height (between ca. 0.5 and 1.5 m). Areas with such a reed height existed only temporarily. Grazing may affect vole abundance in different ways: by trampling, by changing the vegetation structure and/or by influencing the availability of food for voles.

Trampling

Normally, common voles live in burrows in the upper soil and feed above ground. In the case of high densities of ungulates, trampling might well have a serious effect on vole abundance, by destroying burrows and compacting the soil (cf. Heske & Campbell 1991). The risk of trampling is probably higher during wet conditions in winter and therefore higher in year-round grazed areas than in summer-grazed areas. On Lauwersmeer flats, characterised by high water tables in winter and occasionally in summer, common voles are forced to live above the ground for a substantial part of the year. Above-ground, voles live in nests of grass in the vegetation (own observations) and are probably vulnerable to trampling.

Vegetation structure

Herbivores may affect the vegetation structure directly by grazing and trampling, or indirectly by changing the abiotic environment (compacting and thereby salinating the soil, or influencing the level of nutrients (Scherfose 1993)). The vegetation structure is thought to influence vole density (Edge et al. 1995, Peles & Barrett 1996). A uniform, low, vegetation has a negative effect on vole density because it does not offer the voles enough cover. This may have played a role in some of the grazing units. In the summer-grazed study plot in Oostvaardersplassen, vegetation was short-grazed throughout the year and a lack of cover may have been responsible for the low vole densities. In a large part of the year-round grazed study plot, vegetation varied annually between rough grassland in summer and short grassland in winter and early spring. This part of the study plot probably was a favourable habitat for voles in summer, but less so in winter. Flats in Lauwersmeer, characterised by high groundwater tables, undoubtedly need a more structured vegetation for voles to survive in winter than well-drained environments. A lack of cover was probably the main reason for low vole densities in years with a relatively high stocking rate.

In the 14-year study period on former tidal flats in the summer-grazed study plot in Lauwersmeer the vole population did not show the characteristic 3-4 years cycle in density. The absence of this cycle might have been due to large changes in vegetation structure over the years and by high groundwater tables at the flats during winter. Throughout the winter months (October-March) the vole index declined by 95-98% (*n*=2 winters).

Availability of food

Voles (*Microtus* spp.) often fluctuate greatly in number and food availability plays an important role in limiting their numbers (Hansson 1979). Throughout the year, the diet of the common vole in the study areas mainly consists of the green parts of monocotyledons, with the seeds of monocotyledons and dicotyledons playing a role in summer (Hoogeboom, unpublished data).

Grazing affects food abundance and quality for voles in different ways. Through repeated grazing, grass maturation is prevented and growth stimulated, resulting in a higher food quality. However, the prevention of grass maturation suppresses the production of seeds (McNaughton 1979).

Where grassland in the study areas was intensely grazed, seed production, as indicated by inflorescence abundance, was very low. On Lauwersmeer flats, seed production in creeping bentgrass sharply decreased after the introduction of summer grazing (van Eerden et al. 1997). In grassland with a low grazing pressure, seed production appeared to be relatively high during the stage with a moderate reed height. At this stage common voles were relatively numerous. When grassland changed into a closed reed stand, grass-cover - and hence seed production - sharply decreased. It seems probable that the decrease in vole index at this stage can be explained by a decrease in the availability of food.

Effects of vole density and vegetation structure on raptor density

In the year-round grazed study plot in Oostvaardersplassen and the summer-grazed study plot in Lauwersmeer hen harrier and kestrel showed a clear numerical response to vole abundance. The additional effect of vegetation cover was only detected in the summer grazed study plot in Lauwersmeer and not in the year-round grazed study plot in Oostvaardersplassen, but this was probably due to the small sample sizes (number of years). In Lauwersmeer the highest densities of raptors were found at sub-maximum prey densities, where vegetation cover, as indicated by reed height, was somewhat lower. We conclude that the numerical response of vole-feeding raptors to change in vole availability occurred very rapidly. This holds for the hen harrier, which is mainly a winter visitor, as well as for the kestrel, which is a partial migrant and a common breeding bird in the study areas (Cavé 1968, Masman et al. 1988). These results are in accordance with the results of Korpimäki (1994) and Korpimäki & Norrdahl (1989, 1991), who found that the densities of most avian predators in western Finland tracked vole densities rapidly, without obvious time lags.

Maximum raptor densities in the summer grazed study plot in Lauwersmeer were much higher than in the year-round grazed study plot in Oostvaardersplassen, despite lower maximum vole indices in Lauwersmeer. On Lauwersmeer flats, characterised by high water tables in winter and occasionally in summer, common voles are forced to live above the ground for a substantial part of the year. Above-ground, voles live in nests of grass in the vegetation (own observations) and are probably more vulnerable to predation. Additionally, Lauwersmeer flats have a somewhat more open vegetation, because of a more sandy soil.

Conclusions

In the study areas grazing regimes with relatively low stocking rates created a suitable vegetation structure for common voles and subsequently for vole-feeding raptors, for five years at the most. In man-made wetlands, characterised by a low level of physical perturbation, the relevance of grazing as a management tool for encouraging vole-feeding raptors will greatly depend on the potential of grazing to revert tall vegetation to an earlier successional stage.

Vegetation dominated by reeds can be successfully reverted into a vegetation dominated by short grasses by intense grazing (cf. van Deursen & Drost 1990, Vulink et al. 2000, the present study). Subsequently, the regular occurrence of relatively high densities of common voles and vole-feeding raptors can be achieved by creating a cyclic variation in stocking rate. Years with a relatively low stocking rate should be alternated with some years with a higher one. In the first years after lowering the stocking rate, high densities of common voles and vole-feeding raptors can be expected. In Oostvaardersplassen, in 1997 the area with year-round grazing was extended to the entire border zone and stocking rates in this area gradually increased to about 2.4 animals.ha^{-1} in recent years (in 2011 about 350 Heck cattle, 1150 konik horses and 3300 deer (www.staatsbosbeheer.nl)). The area became short grazed for a large part of the year and densities of hen harrier and kestrel decreased to very low levels (less than 0.1 bird-days.ha^{-1}). In spring 2010 ten small exclosures (10x8 m) were erected in the year-round grazed area of Oostvaardersplassen and vole trapping with life traps in October 2010 showed substantial densities of common voles within the exclosures and no voles in the areas surrounding the exclosures nearby (own observations). In the summer grazed study plot in Lauwersmeer the stocking rates remained very low and the reed stands are nowadays gradually being succeeded by willow species. Since 1995 densities of the hen harrier remained low (less than 2 bird-days.ha^{-1}), and the kestrel became an irregular visitor. Higher stocking rates are needed to revert tall vegetation to an earlier successional stage, before vole feeding raptors are able to profit from the abundance of common voles again.

In addition to vole-feeding raptors, red fox (*Vulpes vulpes*), mustelids (*Mustela nivalis* and *Mustela erminea*) and herons (*Botaurus stellaris*, *Egretta alba* and *Ardea cinerea*) are also able to profit from the abundance of common voles. In years with high stocking rates, raptor densities will be lower, and the area will be more suitable for species that prefer a vegetation of short grasses, such as geese (Vulink et al. 2000, Vulink et al. 2010). If the preservation of vole-feeding raptors is to be a goal of the management of large-scale man-made wetlands, the management should include spatial and temporal variation in grazing pressure, following the pattern of partial migration in natural systems such as the Serengeti (e.g. McNaughton & Banyikwa 1995).

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Samenvatting

De lange termijn-effecten van begrazing met landbouwhuisdieren op de talrijkheid van de veldmuis en de daarvan afhankelijke roofvogels in door de mens gemaakte wetlands in Nederland

Verschillende studies hebben het effect van begrazing door wilde herbivoren of landbouwhuisdieren op de talrijkheid van kleine zoogdieren bestudeerd; sommige studies gaan ook in op het effect van begrazing op de talrijkheid van muizenetende roofvogels. In de meeste studies werd gevonden dat de dichtheid van kleine zoogdieren negatief beïnvloed werd door begrazing, terwijl muizenetende roofvogels meestal een numerieke response lieten

zien op de talrijkheids-index van de veldmuis (*Microtus arvalis*). Echter, de meeste studies waren gebaseerd op gegevens uit een korte onderzoeksperiode (1-4 jaren). Omdat kleine zoogdieren vaak grote aantalsfluctuaties laten zien, is er behoefte aan langjarige studies.

In deze studie analyseren we de lange termijn-effecten (3-27 jaren) van begrazing met landbouwhuisdieren op vegetatieontwikkeling, vegetatiestructuur, talrijkheid van de veldmuis en de daarvan afhankelijke muizenetende roofvogels in recent door de mens gemaakte, grootschalige wetlands in Nederland (Lauwersmeer en Oostvaardersplassen).

De bestudeerde wetlands worden gekenmerkt door een laag niveau van dynamiek, waardoor vroege successiestadia (pioniervegetaties, grazige vegetaties) zonder aanvullend beheer snel vervangen worden door latere successiestadia (riet, duinriet en wilgenstruweel). Begrazing werd in de studiegebieden geïntroduceerd toen de vegetatie nog overwegend bestond uit grazige vegetaties.

Onder invloed van een hoge begrazingsdruk ontstond een homogeen korte grazige vegetatie. Begrazing met een lage begrazingsdruk leidde tot een heterogene vegetatie van grazige vegetaties en riet. Na een aantal jaren ontstonden echter scherpe grenzen tussen kort afgegraasde graslanden en gesloten rietvegetaties. Het intermediaire stadium, gekenmerkt door gematigde riethoogtes (tussen 0,5 en 1,5 m) verdween meer en meer.

Hoge dichtheden van veldmuizen in begraasde gebieden waren beperkt tot gebiedsdelen met gematigde riethoogtes. Gebieden met dergelijke riethoogtes bestonden slechts tijdelijk, waardoor de veldmuisdichtheid na enkele jaren afnam.

Muizenetende roofvogels lieten een numerieke response zien op de veranderende veldmuisdichtheid. Behalve veldmuisdichtheid had ook vegetatiestructuur effect op de roofvogeldichtheid. Maximale roofvogeldichtheden werden gevonden bij een suboptimale veldmuisdichtheid, waar de gemiddelde riethoogte wat lager was.

De relevantie van begrazing als instrument voor het beheer van muizenetende roofvogels in deze wetlands is afhankelijk van de potentie van herbivorie om latere successiestadia weer om te vormen in vroegere stadia. Het regelmatige voorkomen van hoge dichtheden van veldmuizen en de daarvan afhankelijke predatoren kan bereikt worden door het instellen van een cyclisch variërende begrazingsdruk. Jaren met een lage begrazingsdruk zouden afgewisseld moeten worden met enige jaren met een hogere begrazingsdruk.

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The raccoon dog (*Nyctereutes procyonoides*) in the Netherlands – its present status and a risk assessment

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Abstract: The raccoon dog (*Nyctereutes procyonoides*) was introduced from East Asia into the former USSR between 1928 and 1957. Since then it has colonised a large part of Europe and is considered an invasive alien species. An earlier paper (Mulder 2012) reviewed the current knowledge about its ecology. This paper deals with its present status in the Netherlands and provides an assessment of its ecological and human health risks. The colonisation of the Netherlands by the raccoon dog started from north-west Germany about 15 years ago. The pattern of colonisation is blurred by the occurrence of individuals escaping from captivity. Up until 2013 'wild' raccoon dogs were probably recorded exclusively in the north-eastern part of the country. This is in accordance with the distribution in Germany. It seems inevitable that the raccoon dog will colonise the whole territory of the Netherlands in the future, maybe with the exception of the islands in the Wadden Sea. Its general impact on biodiversity is expected to be small. Isolated populations of amphibians, however, may be at risk, as may ground breeding birds in marshes. Raccoon dogs may increase the occurrence of diseases and parasites, of which *Trichinella spiralis* and the small fox tapeworm *Echinococcus multilocularis* probably constitute the most important health risks for humans. The options for effectively managing raccoon dogs are limited; only local and intensive measures of control or predation prevention may have the desired effect.

Keywords: raccoon dog, *Nyctereutes procyonoides*, wasbeerhond, risk assessment, distribution, colonisation, the Netherlands, invasive species, management.

Introduction

In October 2007 the Dutch government published its policy on invasive species (Document 20071012-dn-2007-2899.pdf). According to the definition set out in this document an invasive species is an organism which arrives from elsewhere with the aid of humans (by transport or infrastructure) and which is a successful coloniser (by reproduction and population growth). In accordance with the agreements in the Convention on Biological Diversity (Rio de Janeiro 1992) a succession of policies should be applied to control invasive species: prevention of their arrival, eradication when their populations are still small, and isolation and control management when populations have grown too large to eradicate. The intensity of control measures depends on the impact the invasive species is expected to have on biodiversity and human health and safety. In the Netherlands the Invasive Species Team (TIE) of the Ministry of Economic Affairs has the task to advise the Minister on all issues of invasive species. The TIE collects and publishes information, conducts risk analyses for invasive species and recommends measures for the prevention, control and management of such species. The risk assessment for the rac-

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coon dog was published as an extensive report (Mulder 2011) and an earlier paper reviewed the raccoon dog's ecology in Europe (Mulder 2012). The present paper deals with the history and present situation (as of January 2013) of the raccoon dog in the Netherlands, and also contains a concise risk assessment. It ends by discussing the management options.

Two earlier publications have dealt with the raccoon dog in the Netherlands and the possible risks it poses (Oerlemans & Koene 2008, van Dijk & de Koning 2009). However, these publications were based on a limited selection of the literature, and did not contain an evaluation and analysis of the raccoon dog observations in the Netherlands.

Distribution

To evaluate the history and present situation of the raccoon dog in the Netherlands, all records of raccoon dog sightings until 1 January 2013 were collected and screened. This exercise drew on the available scientific literature, hunting journals and various databases (Alterra, Dutch Mammal Society, Telmee.nl, Waarneming.nl, Yvette van Veldhuijsen (a private individual with a keen interest in raccoon dogs), the Royal Dutch Hunters Society and the AAP Foundation, a rescue centre and sanctuary for primates and other exotic animals). Efforts were made to collect previously unreported observations, through appeals in hunting journals and on the internet. Many of the original observers were contacted by telephone or email and questioned. Observers were asked for their experience with wildlife in general, and with red foxes (*Vulpes vulpes*) and badgers (*Meles meles*) in particular. Details of the way of walking and other behaviour of the observed animal were asked for, as well as the colours of the pelt and the relative length of legs and tail. Most observations by hunters and naturalists could be accepted. Many observations by less experienced people were, however, too vague or incomplete to accept as certain or probable. If observers mentioned having thought of a raccoon (*Procyon lotor*) when they observed the animal, this was taken as a positive sign. The vast majority of the records could thus be validated in four categories: 'certain', 'probable', 'possible' and 'not likely'. All records which could not be verified with additional information, were placed in the 'possible' category. The resulting database has been submitted to the National Authority for Nature Data.

Of course the validation of others' observa-

Raccoon dogs deposit their faeces in concentrated latrines along their routes. *Photo: J.L.Mulder.*

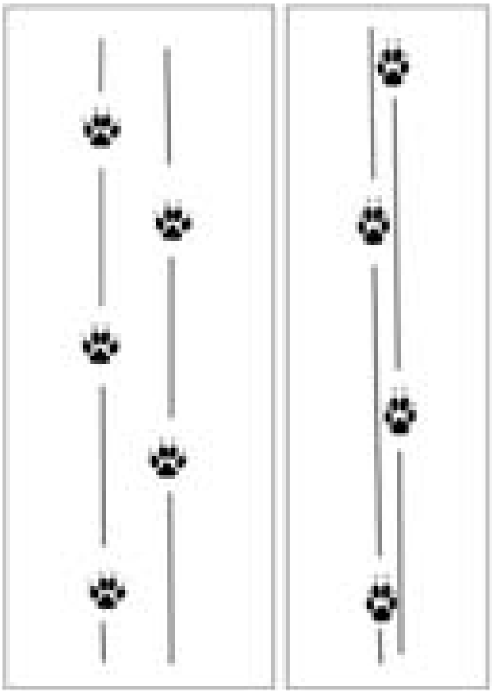

Figure 1. Tracks of raccoon dog (left) and red fox, the former with more rounded prints and with the left and right feet spaced further apart.

Figure 2. Front paw of the raccoon dog, showing the connection between the inner toes. *Photo: Annemarie van Diepenbeek.*

tions is ridden with the subjectivity of the validator. It is impossible to use objective criteria, and the dividing line between the categories cannot be clearly defined. Dead animals pose no problem for identification. Sometimes photographs were made of the animal or the tracks it left in mud or snow, making validation easy for an expert. Footprints were only accepted as proof if they were clearly round and not elongated (as in the red fox), and if the prints of left and right feet were spaced apart instead of placed in almost one line (figure 1). Pictures of prints showing the connection between the caudal part of the two central toes were taken as definitive proof (figure 2). This feature is only visible in sharp prints in mud or clay. Less sharp prints are similar to those of small domestic dogs, and can only be taken as produced by a raccoon dog if the presence of a domestic dog can be absolutely excluded. Until 2013 there were, however, no records based exclusively on prints or other signs (only in connection with an observation of an animal), except for one winter-record of a den with a latrine at five metres distance; this observation was judged ‘probable’.

Escaping raccoon dogs

The first raccoon dog in the Netherlands was observed in April 1981 in the south-east of the country, in the Province of Limburg (Vergoossen & Backbier 1993). The animal did not seem to be very shy, and may well have escaped from captivity or been deliberately set free. That is all the more likely since the first raccoon dogs in north-eastern Germany (at a distance of 600 km from Limburg) were recorded only 15 years earlier, and it was another ten years before the next raccoon dog was recorded in the Netherlands. This second animal was

Table 1. Known cases of escaping raccoon dogs. Sources: Yvette van Veldhuijsen (third and fourth case) and own research.

Date	Place	Number, sex, etc.	How escaped	What happened next
ca 1997	Speuld, Veluwe, children‘s farm	Halfgrown male and female	Unknown	Recovered from a fire wood shed in the next village, Putten (5 km away). After some weeks spent with a private individual they were returned to the children’s farm.
2001	Gangelt (D), Zoo Hochwild Freigehege, on the Dutch/German border, east of Sittard.	Adult male and female	Unknown	Male was killed on an adjacent road on the night of the escape
August 2002	Private house in Enschede	Yearling male and female, brother and sister	Over a garden fence with horizontal wooden beams	Unknown; no road kills shortly thereafter, but one in autumn 2003
September 2002	Private house in Ingen, Betuwe (Gld)	Two yearling animals	One via a small table through an open window. The other some days later by biting through mesh wire	On 12 October 2002 a young female was found as a roadkill near Wijk bij Duurstede. 9 km away on the other side of the river Rhine. During the autopsy it was suspected of being an escapee.
April 2010	Children’s farm Dondertman in Espelo, Holten, Overijssel	Adult male and female, female pregnant	Children left the door of the pen open	Male was killed on the road within 800 m and within a few days, on 26 April 2010.
Autumn 2010	It Schildhus Animal Rescue Asylum, Goengarijp, Friesland (specialising in turtles)	White sterilized adult female and normally coloured neutered adult male. Had been previously kept by private person in Utrecht province.	Were kept in a stable	The white female was shot on 13 November, 5 km away. The male was probably spotted on 11 November, 2-3 km away
ca 10 June 2011	Private person in Egmond-Binnen, Noord-Holland who had already been keeping raccoon dogs for ten years	Two males born in 2010. The third raccoon dog present did not escape.	Over a fence, via the collapsed roof of a dog house	One was killed on the adjacent road 800 m away on 17 June 2011. The other was seen on 20-21 August in Heiloo, around 3 km away, in a chicken coop

found dead by a road east of the town of Groningen (in the north-eastern corner of the Netherlands) in February 1991, and may have been the first ‘wild’ raccoon dog in the Netherlands (Mulder & Broekhuizen 1992).

Raccoon dogs are sometimes kept as pets, indoors as a domestic dog or outdoors in a kennel or behind fencing. There is no commercial farming of raccoon dogs in the Netherlands. They are rather easy to keep, eat virtually anything, hardly make any noise and deposit their faeces in the same spot. Advertisements offering young or adult raccoon dogs are not uncommon in animal journals or on the internet. It is not known how many people keep raccoon dogs in the Netherlands, but it may be in the order of fifty or more. Although a one metre high fence is enough to contain raccoon dogs (Stier 2006), it appears that escaping animals are not rare, mostly through the negligence of the owners. Seven such cases have been documented (table 1), all regard-ing two animals each. In four of these seven cases a road killed raccoon dog was found in the vicinity shortly afterwards, and in one case an animal was shot by a hunter. This last animal certainly was the escapee in question, for it was a white sterilised female. Five of the recovered escapees were found within 0.5 to 9 km and within one month of the time of escape; one was killed on the road on the same night it escaped.

These escaped raccoon dogs may blur the pattern of colonisation in the Netherlands. Isolated observations, far away from the majority of the other observations, may indicate such escapees. Proof, however, is rarely available. Sometimes it is obvious from the behaviour of a raccoon dog that it is an escapee: such individuals show no fear of people: it is possible to come within a few metres, or even less, of them. When such behaviour had been reported, or a connection with a known escape was clear, the record was classified as an 'escapee'.

Observations

After validation of all the observations of raccoon dogs until 1 January 2013 the database contained 173 records (excluding the category 'not likely'), of which 77 were certain, 43 were probable and 53 were possible. In the three categories a total of 11, 6 and 1 records respectively were obvious escapees. Figure 3 shows maps of the Netherlands with all these raccoon dog observations, separated into two periods. The majority of observations were made in the north-eastern half of the country, in the Provinces of Groningen, Drenthe, Friesland and Overijssel, and in the Noordoostpolder, a distribution which accords very well with the species' distribution in Germany (figure 4). Most of the records outside this area (indicated with a dashed line in figure 3) could be escaped animals. The Veluwe area (shown as an oval in figure 3), which is far from the German border, is interesting in this respect. Raccoon dogs started to be reported from the Veluwe relatively early (1993) and have continued to be sighted. From their behaviour, at least three of the observed animals were clearly escaped animals. In addition, several people in the region were known to have raccoon dogs as pets. I therefore consider all the raccoon dog records from the Veluwe (until now) as observations of escaped animals.

Focussing on the north-eastern corner of the country only (with 55 certain, 24 probable and 24 possible observations) and excluding the clear escapees, the pattern of raccoon dog observations over time becomes clear (figure 5). After three isolated observations in the early 1990s, there have been continuous observations of raccoon dogs every year since 2001. In north-east Germany the first raccoon dogs were seen in 1964. That means that the raccoon dog has crossed the distance to the Netherlands in 37 years, at an average speed of 13 km/year. Since 2001 roughly 3 to 8 raccoon dog observations have been recorded in the Netherlands each year, with a maximum of 11-15 in 2006. Most probably however, many raccoon dogs go undetected. The Netherlands apparently has reached the stage of irregular, sporadic but steady observations of raccoon dogs. In other countries this period lasted 20-30 years before the population started to grow exponentially (Mulder 2012). This is in keeping with the apparent lack of reproduction in the Netherlands so far. With their preference for badger setts the pups of raccoon dogs or their signs (latrine) are likely to be easily detected by badger watchers and hunters; up till now no such observations have been reported. Based on the temporal pattern of raccoon dog colonisation in other countries, the exponential growth period may be expected roughly to begin around the year 2025. By 2035 the raccoon dog may be a common inhabitant of the Netherlands.

At present it is unclear how the (probably rather lonely) raccoon dogs behave. Some animals apparently settle for a while in a limited area and are spotted several times. In 2003 a raccoon dog seems to have spent an entire

A

B

Figure 3. All records of raccoon dogs in the Netherlands. Left: until 1 January 2006. Right: between 1 January 2006 and 1 January 2013. The observations from within the oval (Veluwe) are all considered escapees from captivity. The records north-east of the dashed line are mostly considered to be natural colonisers and are used as the basis of figure 5. The records south-west of the dashed line probably all relate to escaped animals, at least before 2006. Black dot: certain; black triangle: probable; open triangle: possible; cross: certainly or most probably an escaped animal.

Figure 4. Distribution of the raccoon dog in Germany, by municipality, according to a repeated inquiry among hunters, by the German project WILD. Black: at least one observation. Grey: no observations. White: no data. Upper map: 2006. Lower map: 2011 (adapted from: http://medienjagd.test.newsroom.de/wild_2011_low_rz_neu.pdf).

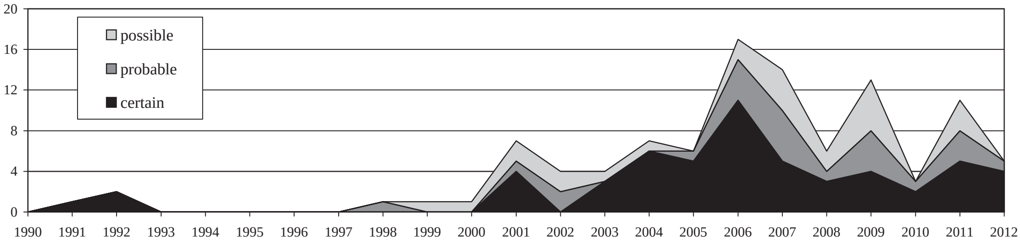

Figure 5. All observations of the raccoon dog in the north-eastern part of the Netherlands (see figure 3) since 1990, grouped in three classes of certainty. Apparent escapees have been excluded.

summer at a tree nursery near Steggerda, Province of Friesland; it was last spotted in November. Another example is a raccoon dog which was seen on 29 May 2011 in Saeftinghe, in tidal salt marshes in the south-west of the Netherlands, far from any other raccoon dog observation. Its tracks in the mud were found at least eight times throughout the summer, for the last time on 25 August (see photo on page ...). Despite many visits to the area, no tracks were found thereafter (personal communication, M. Buise). Other raccoon dogs probably roam the country. On a permanent feeding station (under video surveillance) for wild boar (*Sus scrofa*), badgers and foxes in the Veluwe area, a raccoon dog showed up only once, in December, which suggests that it had not settled there.

Considering the same raccoon dog observations as in figure 5 (for the north-eastern part of the country, excluding escapees) but taking into account only the ‘certain’ and ‘probable’ categories (*n*=79), the methods of observation were as follows: observed in the field (*n*=38), killed on the road (*n*=24), killed by a harvesting machine (*n*=7), shot (*n*=4), caught alive and killed (*n*=2) and drowned (*n*=1). In three cases the details of observation were not exactly recorded. Crops in which raccoon dogs were killed, were beets (*n*=4) and maize (*n*=3). Observations of live animals in the field were often made in the headlights of a car (*n*=11), during lamping (fox shooting with a spotlight, *n*=6) or during daylight hunting (*n*=4). Once (and twice outside the north-eastern part of the country) a raccoon dog was captured on photo by an automatic wildlife camera. Two dead raccoon dogs (one traffic victim, one drowned) appeared to be juvenile; they were collected on 12 and 30 August, in Stiens (Province of Friesland) and Delden (Province of Overijssel) respectively. They most probably are examples of early dispersers, from Germany (see Mulder 2012). Alternatively, they may have been born in the Netherlands; however, if reproduction was already occurring here and there, more juvenile traffic victims, and observations of pups should be expected (see above). Of the few animals which were sexed, ten were males and two were females. Two raccoon dogs were observed together on four occasions (in the years 2001, 2007, 2011 and 2012); the first pair

Footprints of the raccoon dog observed in the salt marshes of Saeftinghe, summer of 2011. The pencil is 10 cm long. The connection between the two front toes is clearly visible. *Photo: Stefaan Thiers.*

Many of the reliable records of raccoon dogs are traffic victims. *Photo: J.L.Mulder.*

were found as traffic victims on consecutive nights, only 50 m apart, the other three were field observations of two animals close to each other.

In conclusion, the raccoon dog is presently regularly but sporadically observed in the north-eastern half of the Netherlands, in accordance with its distribution in Germany. Every year between about three and eight certain or probable observations are reported. Elsewhere it is still very rare, and most of the raccoon dogs recorded in the south-western half of the country probably escaped (or were deliberately set free) from captivity.

Risk assessment of the raccoon dog in the Netherlands

Probability of arrival and establishment

The analysis of the raccoon dog observations in the Netherlands shows that the first observation of a raccoon dog, which probably dispersed from Germany, occurred in 1991. Considering the present colonisation speed of the raccoon dog in Germany, the lack of potential barriers (i.e. mountain ranges) and the presence of suitable habitats in the Netherlands, it is likely that the raccoon dog will continue with its westward colonisation by dispersal and eventually become established in the whole of the Netherlands. Raccoon dogs have a high dispersal and reproduction potential which, in the Netherlands, will not be adversely affected by predators or diseases (see below). The climate matches that of regions where it has already successfully invaded large areas (for instance the north-east of Germany, with an average annual temperature of 9.7 °C, ranging from a mean of 0.8° C in January to a mean of 18.2 °C in July), so the raccoon dog has the potential to be a truly invasive alien species.

Almost all of the Dutch territory is suitable, or even very suitable, for raccoon dogs. Only the built-up areas will probably not be occupied by the species. Research from elsewhere in Europe indicates that resident raccoon dogs avoid settlements and have not been observed to cross villages, even when these were located in the middle of their home range (Drygala et al. 2008). The numerous wetlands in the low-lying north and west of the Netherlands, as well as wet areas in the rest of the country, will probably harbour a dense population of raccoon dogs in the future. Areas of higher ground, that have badger populations and adjoin marshy areas with lots of dense cover (reeds, willow and alder thickets) will probably constitute the optimal habitat, with a maximum of two adults.km^{-2} locally in the most varied landscapes (Kauhala et al. 2006). Less favoured habitats will be large predominantly pine and fir forests and plantations (such as the Veluwe) and large scale agricultural areas (such as most of the Flevopolders). The presence of enough cover, in the form of dense vegetation, is very important for the raccoon dog. Average pre-breeding population density over large suitable areas will probably be between 0.5 and 1.0 adults.km^{-2}.

Impact on native predators

The raccoon dog is an omnivorous, medium-sized predator whose ecology has shared aspects with several native and non-native predators: badger, red fox, polecat (*Mustela putorius*) and American mink (*Mustela vison*). Since the American mink is a non-native species itself, and nothing is known about its relations with the raccoon dog, the species of interest here are badger, red fox and polecat. Impacts can occur as a result of interference competition or resource competition (Pianka 1978). Raccoon dogs rarely seem to directly interfere with badgers (Mulder 2012). Raccoon dogs may kill badger cubs (Sidorovich 2011) but the reverse, badgers killing raccoon dog pups, may be more common (Kowalczyk et al. 2008). Once an adult raccoon dog was observed which had died of wounds on its back most probably inflicted by a badger (Drygala 2009). Indirect interference of raccoon dogs with badgers is suspected to be contributing to the decline of the badger population in central Belarus. Several dead badgers were found in their setts during winter, which apparently had died of suffocation: raccoon dogs often hibernate in badger setts and block almost all the entrances (Sidorovich 2011). In the mild winters in the Netherlands this will be less of a problem. Interference with foxes probably is much more common and both species occasionally kill cubs of the other species. In north-east Germany fox numbers (measured as the number of foxes shot annually) decreased in the first period after the arrival of the raccoon dog, but this effect disappeared later (Zoller 2006). An increase in the infection rate of sarcoptic mange (see below), which is more common in raccoon dogs, might have been responsible for this temporary decline in the fox population. Drygala (2009) concludes that in Europe competition between raccoon dog, red fox and badger might take place, but that it is unlikely that the competition is very severe or leads to a significant decrease of either species.

In northern Belarus, an area with severe winters, the strong increase of the raccoon dog population coincided with a strong decrease in polecat numbers in two study areas, and with a decrease in pine marten (*Martes martes*), red fox and brown bear (*Ursus arctos*) in one of the two study areas. The mechanism behind the impact of raccoon dogs on other generalist predators is thought to be the effective exploitation of available carrion by raccoon dogs in early winter, resulting in a lack of food for the other predators in late winter; a classic example of resource competition (Sidorovich et al. 2000). In the Netherlands such a competition over carrion seems unlikely, except perhaps with the raven (*Corvus corax*). However, some resource competition between raccoon dog and polecat might be feasible, especially with regard to amphibians. Both polecat and raccoon dog, although generalist predators, have a clear preference for amphibians.

Impact on prey species

Raccoon dogs forage while slowly walking, mostly in dense vegetation. They do not 'hunt' like foxes, chasing their prey species. Upon encountering bird nests, they will eat the eggs and chicks, but only rarely the adult breeding bird (Mulder 2012). However, remains of eggs in raccoon dog stomachs are rare in diet studies. According to most authors, the added impact (on top of the impact of native predators such as the red fox) of the raccoon dog on the breeding success of ground nesting birds will probably be negligible. However, solid research into the impact of the raccoon dog on its prey species is still lacking. The predictions in this section are thus mainly based on expert judgment.

Bird colonies in wetlands (e.g., greylag goose (*Anser anser*), black-headed gull (*Croicocephalus ridibundus*)) might be especially vulnerable to raccoon dog predation; as a raccoon dog can possibly destroy many nests in a short time. For the Netherlands, with its many low-lying wetland areas, including many Natura 2000 areas, the most vulnerable species will probably be the purple heron (*Ardea purpurea*), the black tern (*Chlidonias niger*) and the solitary bittern (*Botaurus stellaris*). Although the red fox has already (in recent decades) arrived in most of these wetlands, the raccoon dog may pose an added threat because of its greater readiness to swim. The species mentioned above are possibly at risk and measures to prevent predation by raccoon dogs may be necessary in the future.

The raccoon dog's preference for amphibians may lead to local declines of more or less isolated populations of toads, frogs and possibly newts, for instance in and around cattle drinking ponds in the dryer east and south of the Netherlands. Raccoon dogs also forage on grass snakes (*Natrix natrix*) (Drygala 2009), and might be a threat to isolated populations of this species as well. The common practice of protecting amphibians from being killed on the road in spring, by erecting fences and catching the animals in buckets during the night, may in the future attract the unwelcome attention of raccoon dogs, seeking to gather an easy meal from the buckets (Puffpaff 2008).

Viruses and parasites

The raccoon dog can play a role in the transmission of several infectious diseases, including parasitic diseases, to other species and/or to humans. As far as is known there are no new viruses or parasites imported by raccoon dogs to Europe, however, they may act as a reservoir for several pathogens already present in Europe. Rabies, caused by the classic rabies virus, is an important disease in canids. Until the introduction of the raccoon dog, the red fox was by far the main reservoir and vector of rabies in Europe. That situation has now changed. In Estonia about 50% of wildlife rabies cases were raccoon dogs (WHO 2004 as cited in Kauhala et al. 2007). During the rabies epizootic in Finland in the late 1980s, and later in Poland and the Baltic states, the raccoon dog was the main vector and victim of rabies, accounting for 73% of all reported rabies cases (Westerling 1991, Mól 2005, Kowalczyk 2007, Zienius et al. 2007). Many countries in Western Europe are free of rabies as a result of oral vaccination campaigns. Currently the front of rabies and rabies vaccination campaigns runs from eastern Poland to Croatia (figure 6). In the new situation in Europe, with two main rabies vectors, the vaccination campaigns may not be as effective as before; rabies might persist in the animal community (due to incubation times of up to several months), even if the disease is not spreading in an individual vector species as a result of its low density (Holmala & Kauhala 2006, Kauhala & Kowalczyk 2011). However, Poland has in fact been largely freed from rabies as a result of vaccination campaigns between 2000 and 2010, when raccoon dogs were already as common as foxes. Apparently the oral vaccination campaigns there were as effective for raccoon dogs and foxes together, as they were for foxes alone.

As in other canids, canine distemper caused by canine distemper virus (a paramyxo-virus), an important disease in domestic dogs, has been observed in the raccoon dog. Recently it has become rather widespread in eastern Germany (N. Stier, personal communication). An outbreak of canine distemper in 1991 near Tokyo killed about 70% of the local raccoon dog population (Machida et al. 1993). Canine distemper virus is not known to be zoonotic and thus is harmless for humans, but domestic dogs are susceptible; in fact, wild carnivores around the world may be more likely to be infected by non-vaccinated domestic dogs, than vice versa (http://en.wikipedia.org/wiki/Canine_distemper).

Three parasites are of importance in raccoon dogs: the roundworm *Trichinelle spiralis*, the tapeworm *Echinococcus multilocularis* and *Sarcoptic scabie* var. *vulpesi*, a mite causing Sarcoptic mange. In Finland 53 to 72% of the examined raccoon dogs were infected with *Trichinella* sp. (Mikkonen et al. 1995) and in eastern Germany 5.8% (Thiess 2004). In Finland an association between the density of raccoon dogs and the incidence of infection with *Trichinella* larvae in the European lynx (*Lynx lynx*) has been demonstrated (Oksanen et al. 1998). The role of the raccoon dog as a reservoir of *Trichinella* sp. seems remarkable: where raccoon dogs are common (Finland, Estonia) the prevalence of *Trichinella* in foxes is much higher than elsewhere (Oksanen et al. 1998, Oivanen et al. 2002). The percentage of infected wild boar in north-east Germany also increased in line with the number of raccoon dogs shot each year (Pannwitz et al. 2010). Trichinellosis is also a human disease. However, as a result of the control measures in the meat industry, it is rare in Western Europe (J. van der Giessen, personal communication). Trichinellosis is caused by nematodes of the genus *Trichinella* and the disease results from eating raw or undercooked meat from infected domestic pigs (mostly if they have access to outdoor pens) or game animals. Several carnivorous or omnivorous wildlife species (red fox, raccoon dogs, wild boar, rats, other rodents) can be carriers of *Trichinella*, and if pigs eat their bodies, they become infected. In pigs, clinical symptoms are very rare, but in humans clinical symptoms can occur and are dependent on the dose of *Trichinella* larvae ingested. In worst-case scenarios, the central nervous system and the myocard may be affected, with potentially fatal consequences.

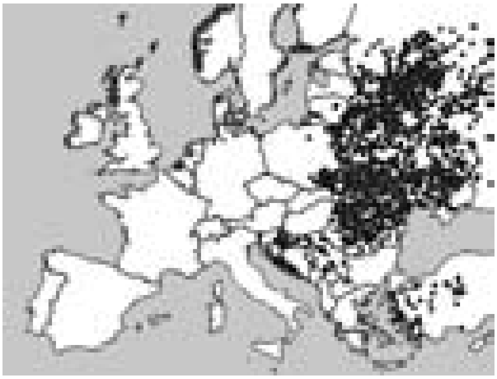

Figure 6. All cases of sylvatic rabies (excluding bats and domestic animals) in Europe in 2012. Map generated with http://www.who-rabies-bulletin.org/Queries/Maps.aspx.

The most important parasite in raccoon dogs (and foxes) is the small fox tapeworm *Echinococcus multilocularis*. This can cause a severe infection in humans, and if not diagnosed and treated properly the infection may lead to death. The parasite is very small (1.2 to 3.7 mm; Faust & Russell 1964) and has no effect on the carnivore carrier, even if it has a high burden of tapeworms. Eggs are excreted with the faeces, thus contaminating the vegetation. Rodents, especially voles, eat the grass and act as secondary hosts. When foxes or other predators consume an infected vole, the life cycle of the parasite is closed. If humans ingest eggs of this parasite, the larval stage develops in internal organs, primarily in the liver. The incubation time is long, 5-15 years, and the route of transmission is as yet unknown. *Echinococcus* eggs are very small and may be ingested via water, contaminated food or direct contact with infected animals, such as a domestic dog.

In Poland and eastern Germany several studies on *E. multilocularis* in raccoon dogs have been conducted (Machnika-Rowinska et al. 2002, Tackmann et al. 2003, Thiess 2004, Schwarz et al. 2011). The percentage of infected animals ranged from 0 to 10.7%, but the studies have been limited in scope and number, so far.

Intestinal parasites such as nematodes, cestodes and trematodes are common in raccoon dogs, as in foxes, and are relatively harmless. Barbu (1972) found an emaciated raccoon dog at the end of spring, with the exceptional number of 1700 trematodes in its intestines.

Sarcoptic mange (*Sarcoptes scabie var. vulpesi*, a mite) is rather common in raccoon dogs (Stier 2006). The mite lives in the skin, causing bare patches, severe itching and, if the infection is severe, may cause death from exposure. Sarcoptic mange sometimes comes in 'waves', decimating the populations of red foxes (Lindström & Mörner 1985) and probably also of raccoon dogs. At other times it is permanently present in low intensities in fox, raccoon dog and other wildlife populations, claiming few victims. Raccoon dogs seem to be more frequently infected with mange than foxes (N. Stier, personal communication). At present mange is rare among red foxes in the Netherlands, and the arrival of the raccoon dog may boost its occurrence (cf. Stier 2006), especially in the fox population, but other species may be affected as well. Humans may be infected by the *Sarcoptes scabei* mite of dogs, and also of foxes, although the mite is not able to finish its life cycle in humans.

Human safety and health risks

Up till now there are no records of raccoon dogs being aggressive towards people. They avoid contact and when cornered keep quiet and can be grabbed easily. However, raccoon dogs can be carriers of diseases and parasites that are harmful to people, i.e. the rabies virus and the fox tapeworm *Echinococcus multilocularis*. Kauhala & Kowalczyk (2011) consider this to be the most severe risk of the raccoon dog's colonisation of Europe. However, rabies has been eradicated in Western Europe and it is highly unlikely that it will return with the arrival of the raccoon dog (see above). The risk of rabies in Western Europe now mainly comes from imported pets (dogs and cats) and from a few bat species (Lina & Hutson 2006). This is said, the rabies control measures in Europe should be scrutinised and, where necessary, reviewed in order to remain successful (Kauhala & Kowalczyk 2011).

The most important health risk constitutes the small fox tapeworm *Echinococcus multilocularis*, of which the raccoon dog is a carrier. Before the arrival of the raccoon dog, the red fox was the only vector of *Echinococcus*. It is unknown whether the raccoon dog will show the same prevalence of *E. multilocularis* in the future as the red fox does now. In the Netherlands the distribution of the small fox tapeworm is restricted to two areas, the north-east and the south-east corners of the country (Giessen et al. 2004a, Giessen et al. 2004b, Opsteegh et al. 2013). It is expected that the distribution of *E. multilocularis* will slowly expand, as a result of the mobility and especially the dispersal of the red fox. Since the indications are that average dispersal distances of raccoon dogs are larger than those of foxes, it is to be expected that the distribution area of *E. multilocularis* will expand slightly faster than with the fox as the sole vector.

Since raccoon dogs will live alongside the existing fox population, the density of potential *E. multilocularis*-carriers will increase and might double in the future. This implies that the infection risk in endemic areas may increase as well. Red foxes spread their faeces diffusely over their whole territory, while raccoon dogs defecate in just a few latrines. Raccoon dogs thus contaminate only a few confined areas of the environment with *E. multilocularis* and pose a, potentially lower risk for human infection than foxes which contaminate a wider environment. The only known method to combat *E. multilocularis,* and diminish the risk for humans, is the (costly) permanent application of anthelminthic baiting, targeting foxes and raccoon dogs. This involves distributing baits containing the worm-killer Praziquantel in the field (Hegglin & Deplazes 2008).

Other risks

The raccoon dog is quite an isolated species in the canid family (Mulder 2012), and hybridisation with other dog species is unknown, even in captivity. There is, therefore, no risk of genetic effects on native species. It is unlikely that raccoon dogs will have a substantial impact, directly or indirectly, on ecosystems as a whole, e.g. by disrupting the existing food webs, maybe with the exception of some small island situations (Mulder 2012). To date, there is no record of raccoon dogs having an economic or social impact in Europe. They are shy and clumsy and avoid the vicinity of people and their infrastructures. Direct damage to property is not known, nor is there competition with economically important animals. They do not climb and do not normally predate on pets or poultry. Raccoon dogs might have some economical impact (although there are no data about this as yet) by eating from commercial crops of low hanging fruit (strawberries, blueberries, blackberries etc.) and maize.

Overall assessment

Several methods have been developed to make a numerical risk assessment for invasive species. Branquart (2007) and colleagues devised a simplified hazard assessment for ecological impacts: the Invasive Species Environmental Impact Assessment protocol (ISEIA). The ISEIA protocol is originally designed for species already established somewhere in Europe, and is therefore the most appropriate in the case of the raccoon dog. In this assessment a number of aspects receive a score of 1, 2 or 3 (for low, medium and high risk respectively) and the sum of scores leads to a classification in one of three categories: A. the 'black list', i.e. high environmental risk; B. the watch list', i.e. a moderate environmental risk and; C. species that are not considered a threat for native biodiversity and ecosystems. Table 2 shows how the raccoon dog scores according to this protocol. According to this protocol, the raccoon dog receives a score of 9 (more details in Mulder 2011), and falls in category B, representing a 'moderate environmental risk'. The main factors responsible for this score are its high dispersion and colonisation potential. It should be noted that the ISEIA protocol does not take the human health aspect into account.

Management

In many countries the year round killing of raccoon dogs is permitted (e.g., Sweden, Norway, Estonia, Latvia, Lithuania, Hungary and Poland). However, in Finland, females with pups are protected in May, June and July, and in Belarus hunting is allowed from 1 October to the end of February (Kowalzcyk 2007, Kauhala & Saeki 2008). In Denmark hunting is not allowed unless there is a negative impact on game animals (Kowalczyk 2007). In Germany the different federal regions (*Bundesländer*) have different rules; in most of them raccoon dogs can be hunted year round. The exceptions are: Niedersachsen and Nordrhein-Westfalen, where adult raccoon dogs are protected from March to August, and Schleswig-Holstein where adult raccoon dogs are protected from March to June; Hamburg, where adult and juvenile raccoon dogs are protected from May to August

Table 2. Scoring the ecological risks of the raccoon dog, according to the ISEIA-protocol.

Aspect	Sub-aspect	Risk	Score	Maximum score
Dispersion potential		High	3	3
Colonisation of high value conservation habitat		High	3	3
Adverse impacts on native species	Predation	Medium	2	2
	Competition	Low	1	
	Disease	Low	1	
	Genetic interaction	Low	1	
Alteration of ecosystem function	Nutrient cycling	Low	1	1
	Physical alteration	Low	1	
	Natural succession	Low	1	
	Food web	Low	1	
Total score				9

and Bremen and Saarland, where raccoon dogs are protected year round (http://www.schonzeiten.de; viewed April 2013).

Kowalczyk (2007) discussed the hunting intensity in Europe: "In Finland, the annual hunting bag varied between 75,000-130,000 in 1998-2003 (Kauhala & Saeki 2004, Kauhala, personal communication), ca. 20,000 in Germany (S. Schwarz, personal communication), 6,000-10,000 in Poland (data of Research Station of Polish Hunting Society in Czempiń), 4,000-5,000 in Estonia, 3,500-4,000 in Lithuania (L. Baltrūnaitė, personal communication), and 2,000 in Latvia. In other countries raccoon dogs are hunted occasionally. (…) Locally, intensive trapping with box and wire traps and hunting with dogs may be methods of raccoon dog eradication. Eradication is, however, difficult, because raccoon dogs, like other canids, tend to increase their litter size when hunting pressure on them is high."

Figure 2 in Mulder (2012) shows the bag record of the raccoon dog between 1994 and 2011 for the whole of Germany. It peeked in the hunting season 2007/08 with more than 35,000 raccoon dogs shot. Since then the increase in numbers seems to have stopped, most probably due to epizootics of canine distemper and sarcoptic mange in the north-east of the country. In the mean time the expansion to western parts of Germany is still continuing. In Germany most raccoon dogs are being shot more or less opportunistically, by hunters waiting near a feeding ground for wild boar or roe deer (*Capreolus capreolus*). Raccoon dogs are also attracted to the bait in feeding pits for foxes, and are shot there as well. Some hunters do target raccoon dogs and use dogs to corner them or to find them in burrows, from which they are dug out. Many raccoon dogs are shot in autumn, when they flee from maize fields during the harvest. When fishponds are being drained for the harvest, they work as magnets for raccoon dogs (and their hunters). As with the fox, box and wire cage traps work for juvenile animals but are ineffective for adults (Stier & Joisten 2006).

Stier (2006) argues that the high hunting bag in eastern Germany may look impressive, but that a two- to three-fold intensification of the hunting pressure would be needed to start reducing the breeding population of the raccoon dog. Rabies, one major natural cause of death has recently disappeared, increasing the raccoon dog's expansion potential; however, another cause is slowly returning: the wolf. In his calculations Stier assumes (rather conservatively) a 300% annual potential increase (2 adults getting 6 pups), and a mortality by hunting of 50% and by other causes (traffic, diseases, old age, etc) of 50%. However, a two- to three-fold intensification of hunting is not feasible and not realistic, and in view of the favourable circumstances in habitat and food availability might still have a limited effect. Stier (2006) advises to address specific possible problems, such as predation of bird colonies, on a local scale. Very intensive control between November (end of dispersal) and April (start of reproduction) is needed, each year, to reduce the spring population in a target area (which should not be too large) in order to achieve the desired results.

One interesting example may illustrate the effect of shooting raccoon dogs. In southern Finland raccoon dogs and other medium-sized predators (red fox, pine marten, American mink) were killed by shooting and capturing in a ‘removal area’ of 55 km^2, in order to study the effects of these predators on the breeding success of ducks. In a similar control area (48 km^2) the numbers of medium-sized predators were not controlled. The experiment lasted five years. Raccoon dogs were killed between 1 August to 31 April each year by volunteer hunters. An index of raccoon dog density was obtained using 50 scent stations each spring. A total of 280 raccoon dogs were killed, i.e. 0.73 to 1.36 individuals/km^2 each year. Notwithstanding this effort, no significant decrease in the raccoon dog index was observed. One reason behind this may be that most raccoon dogs were killed in the autumn and were juveniles that would have died anyway (Kauhala 2004). It is clearly not easy to substantially reduce a population of raccoon dogs.

Options for future management

In all or most of the European countries where raccoon dogs are living today, they have been hunted from the moment of their arrival. Despite the high numbers of raccoon dogs killed by hunters, their expansion has not been halted anywhere and there are no indications that the population density has decreased as a result of hunting. To limit population increase it is necessary to annually remove at least the numbers produced each year in excess of the annual mortality. Despite the large hunting bag this seems to be far from the case in areas where the raccoon dog has become common now (Stier 2006). The ‘usual’ shooting of raccoon dogs will, at the most, decrease their expansion rate a little. Preventing the raccoon dog from establishing itself in the Netherlands, if possible at all, would at least require an effort and professional organisation similar to that established for controlling muskrat (*Ondatra zibethicus*) and coypu (*Myocastor coypus*) (Broekhuizen 2007).

At the moment (February 2013) in the Netherlands the raccoon dog is placed in a category of species that may be controlled according to Clause 67.1 of the Bill on Flora and Fauna (Appendix 1 of the Regulation of Management and Damage Control). This list contains mostly non-native and feral species, but also some species which may be harmful to crops or to flora and fauna. However, firearms can only be used to control these species with the permission of the province. Such permission has only been granted in the Province of Friesland.

All this means that two realistic management options remain:

A. Intensive hunting on a local scale (a few km^2) in places where problems (might) arise, during the months with no dispersal and no reproduction (December – March).

B. Prevention of predation by blocking access for raccoon dogs, for instance by (electric) fencing of colonies of breeding birds or ponds with rare species of amphibians. This method has proved to be effective in the case of the fox (J.L. Mulder, unpublished data).

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Samenvatting

De wasbeerhond (*Nyctereutes procyonoides*) in Nederland – zijn huidige status en een risico-analyse

De wasbeerhond heeft zijn natuurlijke verspreidingsgebied in het verre oosten van Azië. Tussen 1928 en 1957 werden duizenden wasbeerhonden uitgezet in de voormalige Sovjet Unie, voornamelijk ten westen van de Oeral. Van daaruit heeft de soort zich over een groot deel van Europa verspreid. De soort wordt beschouwd als een invasieve exoot, omdat hij door mensen is geïntroduceerd, zich succesvol voortplant en zich verder verspreidt. Het beleid in Nederland met betrekking tot invasieve exoten bestaat uit het schatten van de risico's voor de biodiversiteit met aandacht voor de impact op dier- en volksgezondheid en economie. Een vorig artikel (Mulder 2012) bevatte een samenvatting van de huidige kennis van de ecologie van de wasbeerhond in Europa; het onderhavige artikel geeft een overzicht van zijn huidige voorkomen in Nederland en een analyse van de ecologische, veterinaire en sanitaire risico's.

Bijna vijftien jaar geleden werden in Nederland de eerste uit Duitsland afkomstige wasbeerhonden waargenomen. Het patroon van die kolonisatie wordt verstoord door het regelmatig voorkomen van uit gevangenschap ontsnapte dieren. Tot 2013 werden 'wilde' wasbeerhonden waarschijnlijk uitsluitend aangetroffen in het noordoosten van Nederland: de provincies Groningen, Friesland, Drenthe en Overijssel, plus de Noordoostpolder. Dit patroon sluit aan bij het huidige voorkomen in Duitsland. Er is in Nederland nog geen voortplanting geconstateerd. De huidige fase, met jaarlijks een beperkt aantal waarnemingen, heeft in andere landen 20 tot 30 jaar geduurd. Pas daarna begon de populatie snel toe te nemen.

Het is te verwachten dat de wasbeerhond heel Nederland gaat koloniseren, waarschijnlijk met uitzondering van de Waddeneilanden. Zijn invloed op de biodiversiteit wordt in het algemeen ingeschat als beperkt. Alleen kleine geïsoleerde populaties van amfibieën en grondbroedende vogels in moerasgebieden kunnen gevoelig zijn voor predatie. De wasbeerhond heeft geen nieuwe ziektes meegebracht naar Europa, maar zou wel het voorkomen van

reeds aanwezige ziektes en parasieten kunnen bevorderen: hondenziekte, *Trichinella*, schurft en de vossenlintworm *Echinococcus multilocularis*. De wasbeerhond is ook een potentiële verspreider van hondsdolheid, maar deze ziekte komt tegenwoordig na het uitvoeren van grootschalige orale immunisatie-campagnes niet meer voor. De belangrijkste problemen voor de volksgezondheid worden waarschijnlijk gevormd door *Trichinella* en de vossenlintworm.

De mogelijkheden om wasbeerhonden effectief te bestrijden zijn beperkt. De bestrijding die vanaf het begin van de kolonisatie in Duitsland is toegepast, heeft de verdere verspreiding van de wasbeerhond niet merkbaar verminderd en heeft geen invloed gehad op de voorjaarsstand. Er zijn twee beheeropties voor het tegengaan van mogelijke problemen:

A. Intensieve bestrijding op een lokale schaal (maximaal enkele km^2) waar problemen, zoals predatie, zijn, of te verwachten zijn. Alleen effectief in de maanden zonder dispersie van jonge dieren en zonder reproductie: december tot en met maart.
B. Preventie van predatie door de toegang voor wasbeerhonden te verhinderen, bijvoorbeeld door middel van (schrik-)hekwerk rond kwetsbare plaatsen als poelen met zeldzame amfibieën of broedvogelkolonies.

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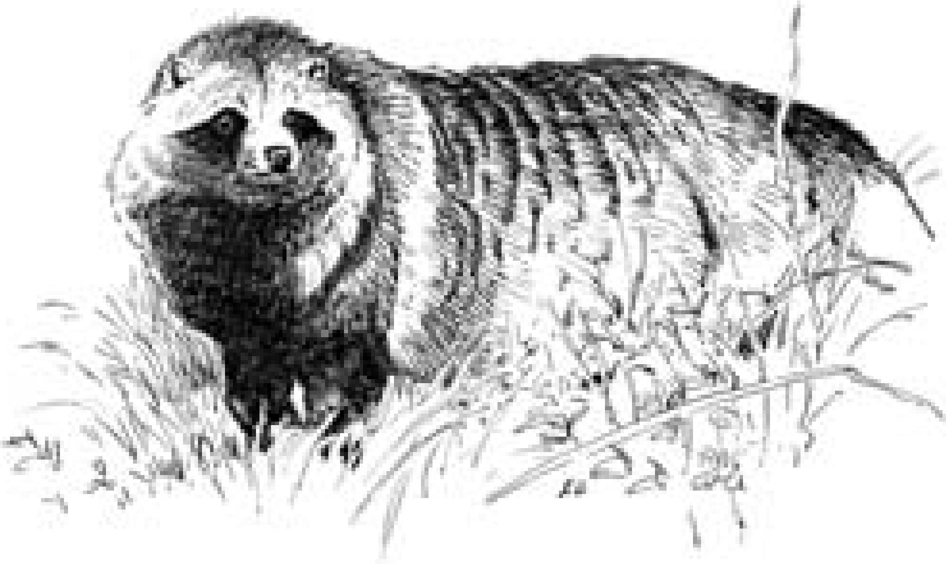

Abundance of harbour porpoises (*Phocoena phocoena*) on the Dutch Continental Shelf, aerial surveys in July 2010-March 2011

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Abstract: The harbour porpoise (*Phocoena phocoena*) is the most abundant marine mammal species in Dutch waters. Nevertheless until 2010 abundance estimates for the entire Dutch Continental Shelf (DCS) were missing. Aerial surveys along designed track lines in July 2010, October/November 2010 and March 2011 provided density and abundance estimates for the DCS. The highest abundance estimate was made in March 2011 (*n*=85,572); approximately three times higher than in July 2010 (*n*=25,998) and October/November 2010 (*n*=29,963). Distribution patterns of porpoises differed between seasons, but a band of higher densities from the Brown Ridge to the Borkumer Reef was visible in all seasons. Calves were mainly seen in July, indicating that porpoises also reproduce in Dutch waters. The total abundance estimate in March 2011 corresponds to 48% of the southern North Sea population, which implies that a large part of the North Sea population resides in Dutch waters during that season. Such high densities may lead to increased conflict with human activities, making the instigation of local management actions more imminent.

Keywords: abundance, aerial survey, distance sampling, harbour porpoise, North Sea, *Phocoena phocoena*, population size.

Introduction

The harbour porpoise (*Phocoena phocoena*) is the most abundant marine mammal species in Dutch waters. After a sharp decline in the first half of the 20th century, the occurrence of harbour porpoises in Dutch waters has increased significantly in the last decades (Camphuysen 2011). This is probably a result of a southward shift in distribution (SCANS 2008). The reasons for this are not clear. A shift in prey species is a likely cause (Camphuysen 2004), although this remains a matter of debate (MacLeod et al. 2007).

Systematically collected data on harbour porpoise abundance and distribution in Dutch waters are scarce. Most data are a by-product of surveys aimed at seabirds, providing information on the relative occurrence and distribution of porpoises. These data were collected during a land-based sea watching scheme, ship-based surveys (Camphuysen & Leopold 1994) and aerial surveys of the Dutch Continental Shelf (DCS) (Baptist & Wolf 1993, Arts 2011).

In the summer of 1994 and 2005, two large-scale dedicated cetacean surveys (SCANS and

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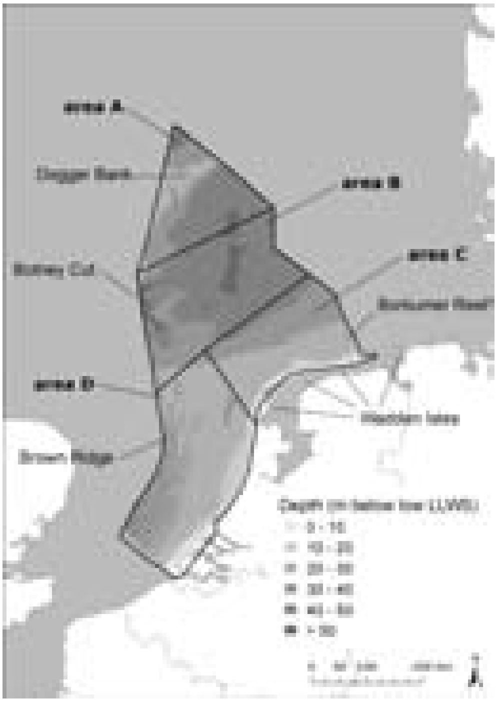

Figure 1. Map of the Dutch Continental Shelf showing the study areas A-D, and some geographical names used in the main text.

SCANS II) resulted in abundance estimates of harbour porpoises in European waters, but the design of the survey areas did not coincide with national borders (Hammond et al. 2002, SCANS 2008). Furthermore, the international survey took place in summer, while in Dutch waters, most porpoises are present in winter and early spring (Camphuysen 2011, Scheidat et al. 2012).

Since May 2008, dedicated aerial surveys were conducted in the Dutch sector of the North Sea. However, in these surveys, the northernmost part was excluded (Scheidat et al. 2012). Hence, an un-biased estimate of porpoise abundance and distribution for the entire Dutch Continental Shelf (DCS) is still lacking. Such baseline data on porpoise abundance and distribution is essential to monitor the effect of human activities and to potentially assess the effectiveness of conservation measures (e.g. Scheidat et al., in press). The objective of this study is to present the results of three aerial surveys in July 2010, October/November 2010 and March 2011, and to estimate the distribution, density and abundance of the harbour porpoise on the entire Dutch Continental Shelf.

Methods

Study area, survey design and data acquisition

Aerial surveys were carried out on the Dutch Continental Shelf, divided into four areas (figure 1): A (9615 km^2), B (16,892 km^2), C (12,023 km^2) and D (20,797 km^2), which were surveyed by aircraft along predesigned track lines. The design of the track lines was parallel in ‘near shore’ areas C and D and zigzag in areas A and B to ensure a representative coverage (figure 2). The direction of transects in areas C and D followed depth gradients in order to minimise potential variance in encounter rate within transect lines caused by depth (Buckland et al. 2001). The zigzag design of the offshore areas aimed at maximising the endurance of the plane and cover as large an area as possible. Additional track lines were surveyed in two smaller areas W1 and W2, within areas D and C, respectively, selected as potential areas for future offshore wind farms (figure 2).

Surveys were conducted with a high-winged twin-engine airplane, the *Partenavia 68*, equipped with bubble windows (allowing observations directly under the plane), flying at an altitude of 183 m (600 feet) with a speed of ca. 186 $km.h^{-1}$ (ca. 100 knots). Every four seconds time and the aircraft’s position were recorded automatically onto a laptop connected to a GPS. Surveys were conducted by a team of three people. Details on environmental conditions were entered in a database by the so-called navigator at the beginning of each transect and whenever conditions changed. Observations were made by two dedicated observers each located at the bubble windows on the left and right side of the aircraft. For each observation, the observers acquired data that were entered in real time into a database by the navigator. Observation data included species (all cetaceans and seals), declination angle measured with an inclinometer from the aircraft abeam to the individual or group, group size, presence of calves, behaviour, swimming direction relative to the transect, detection cue, whether the individual or group was above or below the sea surface when abeam, and reaction to the survey plane. Environmental data included sea state (Beaufort scale), turbidity (assessed by visibility of objects below the sea surface), cloud cover (in octaves), glare (area covered and strength) and subjective sighting conditions. These sighting conditions represent each observer's subjective view of the likelihood that the observer would see a harbour porpoise within the primary search area should one be present, and these conditions could be either good, moderate or poor. Furthermore, a category "not possible to observe" is used. Sighting conditions could differ between the left and right side of the plane.

Surveys were conducted in weather conditions safe for flying operations (no fog or rain, no chance of freezing rain, visibility >3 km) and suitable for porpoise surveys (sea state ≤3 Beaufort).

Data analysis

The survey data were collected using distance sampling techniques (Buckland et al. 2001, Buckland et al. 2004). The collected sightings were used to calculate densities and abundance estimates, and to produce distribution maps. For the former only the surveys of the areas A, B, C and D were used (i.e. the extra survey effort in W1 and W2 was omitted), while for the distribution estimates all data were used. Line-transect distance sampling allows for obtaining estimates of absolute densities, i.e. the number of animals.km^{-2} with the associated 95% confidence interval (C.I.) and coefficient of variation (C.V.; Buckland et al. 2001). Animal abundance in each stratum v (i.e. area A, B, C and D) was estimated using a Horvitz-Thompson-like estimator (Buckland et al. 2001, Buckland et al. 2004) as follows:

$$\hat{N}_v = \frac{A_v}{L_v}\left(\frac{n_{\mathrm{gs}v}}{\hat{\mu}_{\mathrm{g}}} + \frac{n_{\mathrm{ms}v}}{\hat{\mu}_{\mathrm{m}}}\right)\bar{s}_v$$

where A_v is the area of the stratum, L_v is the length of transect line covered on effort in good or moderate conditions, n_{gsv} is the number of sightings that occurred in good conditions in the stratum, n_{msv} is the number of sightings that occurred in moderate conditions in the stratum, $\hat{\mu}_g$ is the estimated total effective strip width (ESW) in good conditions, $\hat{\mu}_m$ is the estimated total effective strip width in moderate conditions and S_v is the mean observed school size in the stratum.

The effective strip width is the distance at which the number of animals detected outside the strip width equals the number of animals missed inside the strip width. The ESW is corrected for the proportion of animals missed on the track line by a factor called g(0). For the current survey the g(0) values obtained in a similar German study (Scheidat et al. 2005, Scheidat et al. 2008, Gilles et al. 2009) were applied. These g(0) values are 0.37 for good conditions and 0.14 for moderate conditions, resulting in effective strip widths of 76.5 and 27 m, respectively.

Group abundance by stratum (areas A, B, C and D) was estimated by:

$$\hat{N}_{v(\mathrm{group})} = \hat{N}_v / \bar{s}_v.$$

Total animal and group abundances of the entire study area were estimated by

$$\hat{N} = \sum_v \hat{N}_v \text{ and } \hat{N}_{(\mathrm{group})} = \sum_v \hat{N}_{v(\mathrm{group})}$$

respectively. Densities were estimated by dividing the abundance estimates by the area of the associated stratum. Mean group size across strata was estimated by

$$\hat{E}[s] = \hat{N} / \hat{N}_{(\mathrm{group})}.$$

Coefficients of variation (C.V.) and 95% confidence intervals (C.I.) were estimated by a non-

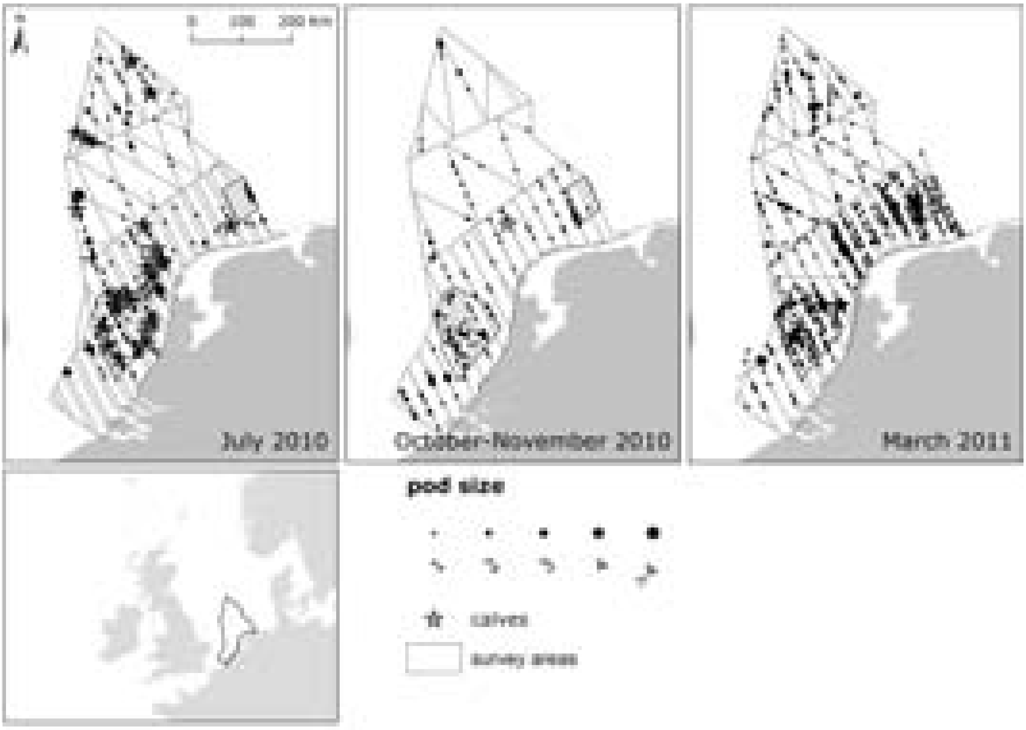

Figure 2. Survey effort in good or moderate sighting conditions on at least one side of the plane (on and off track line) with all sightings of harbour porpoises, including navigator sightings. Stars indicate groups with calves/neonates.

parametric bootstrap (999 replicates) within strata, using transect segments as the sampling units. The variance due to estimation of ESW was incorporated using a parametric bootstrap procedure assuming the ESW estimates to be normally distributed random variables. More details on this method can be found in Scheidat et al. (2008, 2012).

Distribution maps

Distribution maps are presented in two different ways: 1. Uncorrected sightings, and 2. Densities corrected for survey effort and subjective sighting conditions per 1/9 ICES grid cell. Densities are represented spatially on a 1/9 ICES grid. This grid has latitudinal rows at intervals of 10' and longitudinal columns at intervals of 20'. Within the DCS, this corresponds to approximately 20x20 km grid cells, with areas ranging from 388 to 409 km^2.

Densities per 1/9 ICES grid cell were calculated by dividing the total number of animals observed during good and moderate conditions by the total surveyed area. The surveyed area is the distance travelled multiplied by the effective strip width (ESW). Grid cells with low effort, such as grid cells extending outside the borders of the surveyed area (e.g. the Wadden Sea), tend to be less reliable. Therefore, grid cells with an effort less than one km^2 were omitted from the maps (but used for the abundance estimates).

Results

Effort and sightings of harbour porpoises

The July 2010 and March 2011 surveys had the best coverage with both around 6000 km

Table 1. Total survey days, effort, sighting conditions (G – good, M – moderate, P – poor, X – not possible to observe) and harbour porpoise sightings during the three aerial surveys. Calves are included in the number of animals.

		Sighting conditions (%)			Porpoise sightings (n)		
Survey	Effort (km)	G	M	P / X	Sightings	Individuals	Calves
July 2010 (5,6; 8-11; 18-20 July)	6040	35	35	30	263	330	26
October/November 2010 (12-14 Oct; 19, 21, 24 Nov)	4028	12	76	11	137	163	0
March 2011 (18,19; 21 – 27 March)	5945	29	62	9	684	743	2
Total	16,013				1085	1236	28

on effort (table 1). In these months, two sets of track lines in areas A-D plus the extra track lines in W1 and W2 could be surveyed (figure 2). Survey conditions were more adverse in October and November (shorter days, unstable weather) leading to lower survey effort, but one set of track lines could be completed in areas A-D and in the W1 and W2. In total, 1085 sightings of 1236 harbour porpoises were collected. Calves were sighted during the July survey and the March survey, consisting 7.9% and 0.3% of the sighted individuals, respectively. During the October survey, one calf was recorded by the navigator (navigator sightings are not included in the analysis of abundance). The 'calves' in March were big neonates.

Average group size for all surveys combined was 1.14 animals. Average group size was 1.09 (C.V. 0.31) animals in March, 1.25 (C.V. 0.52) in July, and 1.19 (C.V. 0.37) in October/November. The largest group size observed was a pod of eight animals in July. In all seasons over 80% of the sightings consisted of single animals. In summer and in autumn a larger proportion of the sightings consisted of two or more animals compared to early spring.

Density and abundance of harbour porpoises

Table 2 gives an overview of density (animals.km^{-2}) as well as abundance (number of animals) per survey area and survey period. The overall density was similar for the summer (July) and the autumn (October/November) survey; 0.44 and 0.51 animals.km^{-2} respectively. Density was about three times higher during the March survey with 1.44 animals.km^{-2}.

The total numbers of harbour porpoises on the DCS (areas A-D) were estimated at 25,998 (C.I.: 13,988-53,623) and 29,963 (C.I.: 16,098-59,011) animals in summer and autumn respectively. The abundance in March comprised 85,572 animals (C.I.: 49,324-165,443, table 2).

Distribution of harbour porpoises

Figures 3-5 show densities of porpoises (animals.km^{-2}) per 1/9 ICES grid cell. In summer (July), higher densities were observed near the Brown Ridge, the Borkumer Reef, and around the Botney Cut – Dogger Bank, near the UK border. The band of higher densities running from the Brown Ridge to the Borkumer Reef was also visible in the autumn surveys (October/November) and the spring surveys (March). In autumn the offshore density in area B was lower and porpoises were more evenly distributed than in the other two survey periods. In spring, the overall density was much higher. In that period, the densities in areas B and D almost tripled, while area C showed an even stronger increase. The highest density was found in area C, north of the Wadden Isles (figure 6). The high density area in the Dutch part of the Borkumer Reef

Figure 3

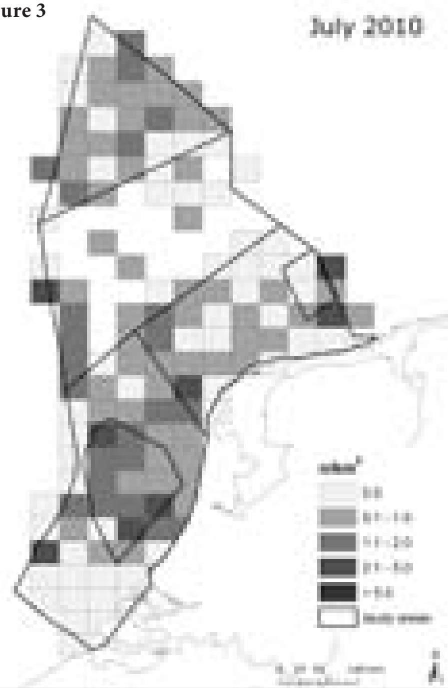

Figure 4

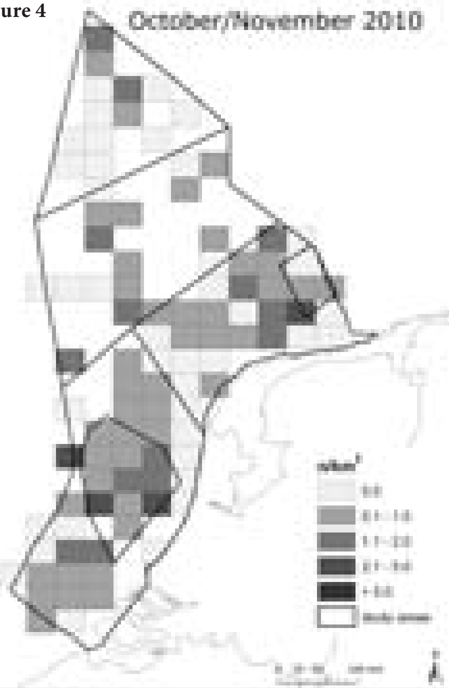

Figure 5

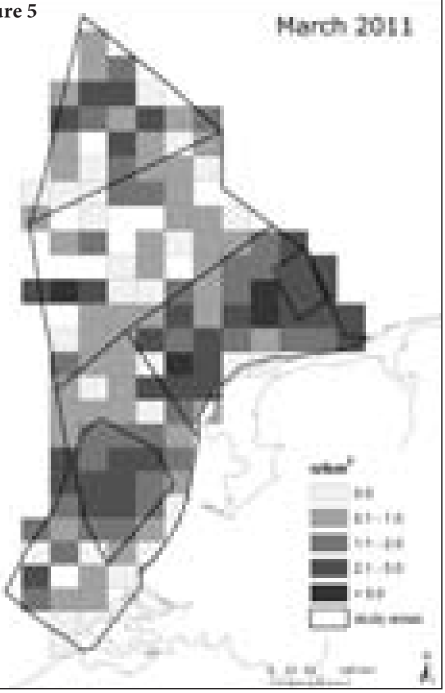

Figure 3. Summer density distribution of harbour porpoises (animals.km^{-2}) per 1/9 ICES grid cell, July 2010. Grid cells with low effort (<1 km^2) are not presented.

Figure 4. Autumn density distribution of harbour porpoises (animals.km^{-2}) per 1/9 ICES grid cell, October/November 2010. Grid cells with low effort (<1 km^2) are not presented.

Figure 5. Spring density distribution of harbour porpoises (animals.km^{-2}) per 1/9 ICES grid cell, March 2011. Grid cells with low effort (<1 km^2) are not presented.

Table 2. Estimates of density and abundance of harbour porpoises from aerial surveys conducted on the Dutch Continental Shelf in July 2010, October/November 2010 and March 2011. Estimates are given with the associated 95% Confidence Interval (C.I.) and Coefficient of Variation (C.V.).

July 2010

Survey Area	Density (animals.km^{-2})	(95% C.I.)	Abundance (*n* animals)	(95% C.I.)	C.V.
A	0.396	0.181 - 0.849	3806	1738 - 8165	0.404
B	0.477	0.212 - 1.058	8055	3589 - 17,872	0.416
C	0.336	0.046 - 0.890	4039	553 - 10,701	0.622
D	0.484	0.208 - 1.056	10,098	4341 - 22,024	0.403
Overall	0.438	0.236 - 0.903	25,998	13,988 - 53,623	0.336

October / November 2010

Survey Area	Density (animals.km^{-2})	(95% C.I.)	Abundance (*n* animals)	(95% C.I.)	C.V.
A	0.391	0.117 - 0.872	3763	1124 - 8384	0.461
B	0.573	0.298 - 1.157	9679	5035 - 19,543	0.352
C	0.683	0.287 - 1.610	8216	3451 - 19,351	0.459
D	0.398	0.212 - 0.733	8304	4431 - 15,296	0.317
Overall	0.505	0.271 - 0.994	29,963	16,098 - 59,011	0.332

March 2011

Survey Area	Density (animals.km^{-2})	(95% C.I.)	Abundance (*n* animals)	(95% C.I.)	C.V.
A	1.029	0.522 - 2.144	9890	5018 - 20,618	0.386
B	0.908	0.521 - 1.791	15,331	8795 - 30,249	0.312
C	2.982	1.645 - 5.806	35,850	19,772 - 69,808	0.325
D	1.174	0.658 - 2.389	24,501	13,726 - 49,833	0.344
Overall	1.441	0.830 - 2.786	85,572	49,324 - 165,443	0.316

extended to the German part. Low densities were found in close proximity of the mainland coast and in the southern part of area D.

Discussion

Seasonal patterns

The results show distinct differences in abundance and distribution of harbour porpoises between surveys; in summer and autumn similar numbers and densities occurred, while in March the numbers almost tripled. This pattern fits the general seasonal occurrence as seen along the Dutch coast during systematic land-based observations of seabird migration and marine mammals (Camphuysen 2011) and previous aerial surveys (Scheidat et al. 2012). These land-based observations show that harbour porpoises are present in coastal waters throughout the year. Peak numbers are observed from winter to early spring (December-March), after which the numbers drop. Observations in June are relatively scarce, but the numbers slightly increase from July onwards (Camphuysen 2004, Camphuysen 2011). In the Belgian part of the North Sea, harbour porpoises are most abundant from February to April, whereas lower numbers tend to occur offshore in the rest of the year (Haelters et al. 2011). French strandings data also indicate a higher occurrence of porpoises in the Channel during the

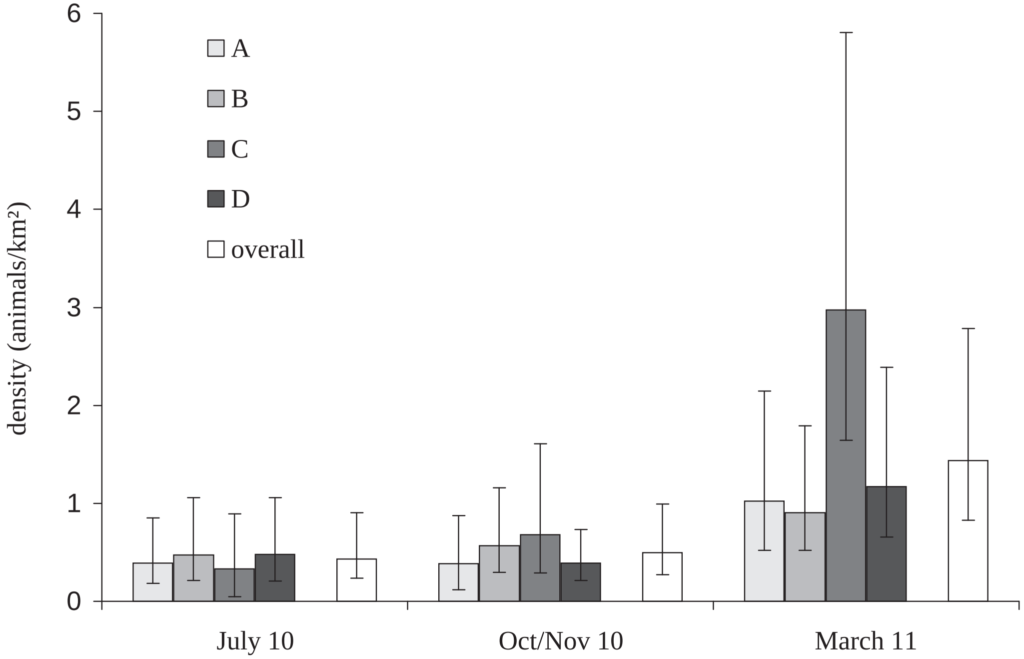

Figure 6. Density of harbour porpoises during the three surveys in the survey areas A-D and the entire study area. The black bars show the 95% Confidence Interval of the estimates.

winter months (Jung et al. 2009). In the German North Sea bordering Dutch waters (area C) the aerial surveys show the highest densities in spring. Further north along the German coast numbers peak in May and June (Gilles et al. 2009). Still further north, along the Danish west coast, porpoise densities are highest from April to August, with a peak in August (data for June-July are lacking however) (Teilmann et al. 2008). In the western North Sea porpoise numbers peak in April off south-eastern England, and later further north (August; Evans et al. 2003). These observations suggest a northward summer migration from the Channel, Belgium and The Netherlands to Danish and British waters and a southward migration in autumn.

Comparison of densities

In 2010-2011, the estimated densities of harbour porpoises in Dutch waters ranged from 0.34–2.98 animals.km^{-2} (table 2). These densities are in the same order of magnitude (0.33-2.53 animals.km^{-2}) as obtained during comparable studies in adjacent waters in Germany (Thomsen et al. 2006, Gilles et al. 2009), and in Belgium (Haelters et al. 2011), and in the relevant survey blocks during the large scale SCANS II survey (SCANS 2008). Data from Gilles et al. (2009) reveal that the highest densities in the entire German North Sea Exclusive Economic Zone were reached in spring with an overall density of 1.34 animals.km^{-2} and a density of 0.85 animals.km^{-2} in the area closest to the Dutch border (East Frisia). The SCANS-II survey (SCANS 2008) showed a porpoise density for SCANS-block H in the southern North Sea of 0.36 animals.km^{-2} in June-July 2005. In the adjacent SCANS-block B (the Channel) a density of 0.33 was estimated for 2005. This density corresponds well with the 0.48 animals.km^{-2} estimated for the Dutch survey area D in July, which overlaps with SCANS-block B.

Table 3. Comparison between density and abundance estimates obtained in the same areas and months (2008–2010) using results from the current study as well as from Scheidat et al. (2012). Estimates are given with the associated 95% Confidence Interval (C.I.) and Coefficient of Variation (C.V.).

Area	Period	Density (animals.km^{-2}) (95% C.I.)	Abundance (95% C.I.)	C.V.
October-December				
Area C	Nov 2008	1.020 (0.34 – 2.10)	12,227 (4038 – 25,285)	0.42
	Oct/Nov 2010	0.683 (0.29 - 1.61)	8216 (3451 – 19,351)	0.46
Area D	November/December 2009	1.511 (0.91 – 3.08)	31,515 (18,976 – 64,157)	0.32
	October/November 2010	0.398 (0.21 - 0.73)	8304 (4431 – 15,296)	0.32
February-March				
Area B	March 2010	0.660 (0.28 – 1.45)	11,141 (4692 – 24,560)	0.42
	March 2011	0.908 (0.52 - 1.79)	15,331 (8795 – 30,249)	0.31
Area C	March 2010	1.107 (0.48 – 2.49)	13,309 (5819 – 29,918)	0.44
	March 2011	2.982 (1.65 - 5.81)	35,850 (19,772 – 69,808)	0.33
Area D	February/March 2009	1.468 (0.78-2.70)	30,534 (16,265 – 56,161)	0.33
	March 2010	2.007 (0.82 – 4.04)	41,878 (17,145 – 84,302)	0.39
	March 2011	1.174 (0.66 - 2.39)	24,501 (13,726 – 49,833)	0.34

In March 2011, both Belgian and German waters were surveyed simultaneously with the Dutch surveys, thus allowing a direct comparison of the corresponding densities. The estimated densities of harbour porpoises were high, with 2.53 animals.km^{-2} and 2.09 animals.km^{-2} in Belgian and German waters respectively. These densities correspond well with the maximum SCANS II density of 2.98 animals.km^{-2} in area C, whereas the density in area D of 1.17 animals.km^{-2} was lower.

Inter-annual variability

Aerial surveys in the DCS using the same methods as the present study have also been carried out from May 2008 to March 2010. In February-May, August, November and December 2008-2010 10,557 km were covered on effort during 16 survey days (Scheidat et al. 2012). Within these surveys it was never feasible to cover the complete DCS, but a comparison per area is possible (table 3). Estimated densities and abundances for the winter months (combined surveys from October to December) were similar for area C in 2008 and 2010 (table 3). For area D the estimates in 2009 seemed higher than in 2010. One reason might be that the difference in timing of the surveys (i.e. November/December versus October/November) could make a big difference in local density due to migration. When comparing March surveys, densities in the offshore area B were similar in 2010 and 2011, whereas area C showed an increase in density in 2011. Density in area D has been fairly stable in 2009, 2010 and 2011 - all estimates lie within the confidence intervals of each other - but showed a possible decrease from 2010 to 2011. The total abundances for areas B, C and D showed similar numbers (66,328 animals in 2010 and 75,682 animals in 2011), suggesting a shift in distribution between these two years, with more animals present in the central-eastern DCS (i.e. area C) in 2011 than in 2010.

Reproduction

In recent years, stranding records of harbour porpoises along the Dutch and Belgian coast showed increasing numbers of neonates in late summer (e.g. Haelters & Camphuysen 2009). These strandings are assumed to reflect births in coastal waters. Figure 2 shows the locations where calves were seen in Dutch waters, which were clearly not restricted to the coastal zone. The observations of (small) calves in July suggest that harbour porpoises reproduce in Dutch waters. Sexually mature female porpoises can give birth to one calf each year or every second year (Gaskin et al. 1974). Given a gestation period of 10-11 months (Gaskin et al. 1974, Addink et al. 1995, Lockyer 2003), this means that mating will take place shortly after parturition. Hence, areas with calves are also important reproduction areas. Based on the size of the foetus in by-caught porpoises, Börjesson & Read (2003) estimated the mean conception date in the North Sea to be 25 July (± 20.3 days) and Gaskin et al. (1974) predicted a mean birth peak from the end of May till the end of June. In the German Baltic and North Sea the majority of births take place in May-July, with the first births in March (Hasselmeier et al. 2004). This is in line with the virtual lack of sightings of calves in March 2011 and October/November 2010, and the higher number of calves seen during the July 2010 surveys.

Management implications

Harbour porpoises in the Atlantic can be divided in several populations or management units (MUs). Evans et al. (2009) assessed these for the north-eastern Atlantic. Based on Danish telemetry data (Sveegaard et al. 2011) and other available data (e.g. genetics), they concluded that the North Sea should be divided into two MUs along an arbitrary line running NNW–SSE from northern Scotland to Germany-Denmark. The Dutch porpoises would belong to the MU south of this line: the south-western North Sea and the eastern Channel MU. The boundaries of this MU are not well defined, but it lies within the SCANS II survey blocks V, U, H and B (SCANS 2008). An abundance estimate for this MU is lacking, but given the smaller area it has to be less than the sum of the estimates in the survey blocks V, U, H and B during SCANS II (SCANS, 2008). Based on SCANS II, the estimated number of porpoises in this management unit is less than ca. 180 000 animals. Assuming that the population from the south-western North Sea and the eastern Channel MU stays within this area throughout the year and that the population size did not change much since 2005, at least 14% of this population was present in Dutch national waters in July 2010. In March 2011 this proportion increased to at least 48%. At that time The Netherlands harbour a substantial proportion of the porpoise population in the southern North Sea and the eastern Channel. This emphasises the importance of living up to the commitment the Netherlands have within the EU Habitats Directive to achieve and maintain a favourable conservation status for the harbour porpoise in Dutch waters. First steps in this process have been assessing total numbers occurring in Dutch waters (this paper) and the development of a Dutch Harbour porpoise conservation plan (Camphuysen & Siemensma 2011).

Conclusions

By correcting for biases in the detection probability, this study provides the first un-biased estimate of the harbour porpoise population size in Dutch waters and how it differs between three seasons. The results show that there is a strong seasonal variation in density and that there are areas with higher densities of harbour porpoises in Dutch waters. In March 2011, high densities were found in the whole DCS, except for the southernmost parts and a narrow strip in close proximity of the mainland coast. In July, high densities were found near the Brown Ridge, Botney Cut-Dogger Bank and Borkumer Reef. In October, the distribution seems more spatially homogeneous. Mother-calf pairs were mostly sighted in July, off the coast of the mainland (area D). Since these patterns may show (large) variability, repeated surveys will be necessary to ascertain if the established patterns are consistent within and between years. These surveys could be primarily aimed at surveying high density areas, but as the exact locations of these may vary over time, repeated DCS wide surveys should provide the necessary background information.

As harbour porpoises are a wide ranging species, larger scale multi-season surveys in cooperation with adjacent countries would allow a better understanding of the movements and habitat use of porpoises in the southern North Sea. Using such data in combination with an overarching spatial model including environmental parameters, would provide information on distribution and ultimately on habitat preferences. This information is necessary to develop adequate management and protection measures in relation to offshore activities for harbour porpoises in the future.

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Samenvatting

Voorkomen van bruinvissen (*Phocoena phocoena*) op het Nederlands Continentaal Plat: vliegtuigtellingen in de periode juli 2010 - maart 2011

De bruinvis (*Phocoena phocoena*) is de algemeenste zeezoogdiersoort in Nederlandse wateren. Desondanks waren er tot 2010 geen aantalsschattingen beschikbaar voor het Nederlands Continentaal Plat (NCP). In juli 2010, oktober/november 2010 en maart 2011 werden vliegtuigtellingen langs vooraf ontworpen *track lines* uitgevoerd waardoor het mogelijk was dichtheden en aantalsschattingen van bruinvissen op het NCP te berekenen. De hoogste aantallen werden in maart 2011 (*n*=85.572) gevonden, ongeveer drie keer zo veel als in juli 2010 (*n*=25.998) en in oktober/november 2010 (*n*=29.963). Het verspreidingspatroon verschilde per telperiode, maar gedurende alle telperioden waren hogere dichtheden aanwezig in een strook tussen de Bruine Bank en de Borkumse Stenen. In juli werden kalfjes gezien, hetgeen een indicatie vormt voor het feit dat dat bruinvissen zich in Nederlandse wateren voortplanten. De aantalsschatting voor maart correspondeert met 48% van de populatie in de zuidelijke Noordzee; een groot deel van de Noordzeepopulatie verblijft daarmee in die periode in Nederlandse wateren. De gevonden hoge dichtheden kunnen leiden tot een toename in conflicten met menselijke activiteiten. Het nemen van beleidsmaatregelen om deze conflicten te voorkomen of te mitigeren wordt hierdoor urgent.

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Harbour porpoises (*Phocoena phocoena*) in the Marsdiep area, the Netherlands: new investigations in a historical study area

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Abstract: The harbour porpoise (*Phocoena phocoena*) returned in Dutch waters in the late 20th century after a near-absence of approximately three decades. Inspired by historical studies of harbour porpoises in the Marsdiep area (western Wadden Sea), mainly in the 1930s and 1940s, a study was initiated in the same area in 2011 to see if porpoises are now as common as previously recorded and if future, more detailed investigations would be worthwhile. We investigated the current spatial and temporal distribution of harbour porpoises using a combination of visual observations and acoustic recordings. In 2010 and 2011, harbour porpoises were most abundant around mid-March, and disappeared in April. They were most frequently observed off Texel, with slightly lower numbers of visual detections along the dike of Den Helder and at Huisduinen. Relatively high abundances were recorded around high tide at most observation sites, but particularly so off Texel. The observed abundance of porpoises in spring in the Marsdiep area, and the apparent influence of tidal currents on spatial patterns in sightings frequencies, is such that the Marsdiep area could provide rather unique, future study opportunities for harbour porpoises in the wild.

Keywords: Cetacea, *Phocoena phocoena,* Marsdiep, Wadden Sea, tidal rhythm, foraging habitats, sightings, recordings, history, Netherlands.

Introduction

The harbour porpoise (*Phocoena phocoena*) returned as an abundant, indigenous species in Dutch waters in the late 20th century, after a virtual absence of three decades (1960-1990; Camphuysen 2004, Camphuysen & Peet 2006, Camphuysen 2011, Scheidat et al. 2012). Up to the late 1950s, harbour porpoises were common in Dutch coastal waters. The only area where studies were conducted prior to the disappearance from Dutch waters was the Marsdiep, a sea inlet between Den Helder and Texel in the western Wadden Sea. Verwey (1975) reported sightings, documented behaviour and prey preferences and described seasonal patterns of cetaceans in the Marsdiep area between 1931 and the early 1970s.

The Marsdiep area is unique in the sense that the entire area can be overseen from land, and that its complex bathymetry and oceanography offers a rare opportunity to study wild cetaceans, undisturbed, in a variety of marine

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habitats. Due to its proximity to the Royal Netherlands Institute of Sea Research (NIOZ), in and around the Marsdiep, several oceanographic and biological sampling programs are established. The historical and detailed dataset on the occurrence of harbour porpoises and common bottlenose dolphins (*Tursiops truncatus*) collected by the Zoological Institute just prior to and following the Second World War served as a reference (Verwey 1975).

Following the return of harbour porpoises in the early 21st century, we felt that a resumption of studies in the Marsdiep area (using modern observation techniques) could yield much needed information on habitat characteristics, seasonality and behaviour of harbour porpoises in Dutch coastal waters. We designed a feasibility study which included a systematic porpoise observation programme. A team of observers manned several, strategically chosen observation sites to assess spatial patterns in harbour porpoise abundance within the tidal inlet (visual observations). The long-term seawatching programme at Huisduinen was continued and intensified to provide a recent background of seasonality in the area. Acoustic techniques were deployed to supplement the information derived from sightings. The objective of this study was to investigate if harbour porpoises within the Marsdiep area are currently abundant enough to warrant a more complete ecological study in future years. We collected sightings (with corrections for observer effort and environmental conditions) during the period of peak abundance in Dutch coastal waters and beyond (February-April 2011), and documented site-specific patterns in abundance together with aspects such as tidal phase, time of day, shipping traffic, presence and abundance of seals, and the occurrence of multi-species feeding frenzies of seabirds and marine mammals.

Historical abundance

Verwey's (1975) studies comprised descriptions of sightings and of the behaviour of the animals observed in the period 1931-1940 and 1945-1973 along the shores of the Marsdiep near Den Helder, with numbers being much higher in the earlier period. Jan Verwey and co-workers from the Zoological Station in Den Helder (now NIOZ, Texel) reconstructed the seasonality in occurrence of harbour porpoises, and described interactions with common bottlenose dolphins and their (presumed shared) prey (Verwey 1975). The animals were observed mostly during bicycle rides along the dike, from the ferry to Texel, or occasionally from the institute's research vessel, i.e. they were opportunistic sightings rather than systematic observations. The last sightings of significance were reported by Dudok van Heel (1960), who recorded 40-50 "*Phocaena*" in Texelstroom, the sea inlet just to the southwest of the Marsdiep area. Since then, few sightings were recorded until the late 1990s (Camphuysen 2004). The information on seasonal trends was only based on data gathered between 1934 and 1939, when porpoises were still abundant. The more systematic observations were made between the harbour of Den Helder and the lighthouse of Huisduinen (along a dike), and were given as "monthly totals", where a month comprised circa twelve observation days (during fine weather). From these historical data and other publications we could summarise the historical abundance of harbour porpoises in Dutch coastal water (with emphasis on the Marsdiep area) as follows:

Before World War II, harbour porpoises were common in Dutch coastal waters (Weber 1922, van Deinse 1925). Numbers in the Marsdiep area were very low from February/March to May, increased in early summer (May or June-July) and rather high numbers were seen in winter (November-December to January-February; once in March; Verwey 1975). Verwey's observations contradict descriptions of the seasonal abundance of harbour porpoises in nearshore waters along the mainland coast of the Netherlands by other authors, who referred to the summer as a period with the highest numbers (Heinsius 1914, van Deinse 1925). IJsseling & Scheygrond (1943) noted that most strandings in the Netherlands occurred in the summer months (notably in August), and many stranded animals were apparently neonates (newborns; van Deinse 1925). Harbour porpoises in the Marsdiep area were assumed to follow the tidal current (entering during flood, leaving during ebb; Verwey 1975). These suggestions were supported by only two observations. Some porpoises were observed in extremely shallow waters in the Marsdiep area: in small creaks, and over sand banks at high tide that were exposed at low tide. In 1939, the first signs of a decline in numbers were noted (Viergever 1955). During World War II, it was impossible to collect data, but directly following the war numbers seemed to have declined even further. Verwey (1975) stated that this decrease took place around 1945, but the decline became more obvious in the 1960s. Factual data are scarce, unfortunately, and assessments of (effort corrected) trends in relative abundance cannot be reconstructed from this material.

Methods

The Marsdiep forms an inlet between the Wadden Sea and the North Sea, through which half of the total body of water in the Dutch Wadden Sea passes twice a day, equating to one billion m^3 of water (Zimmerman 1978). This movement has an average flow rate of 1 m.s^{-1}. This dynamic estuary has a highly variable bathymetry with depths ranging from 1 up to 45 meters (figure 1). The tide in this area can be classified as a semi-diurnal pattern with a cycle of 12.25 hours (Dronkers 1964). The tide is asymmetric in this area with a relatively long-lasting high tide. These high tides can consist of double-headed high tides or a so called “agger”, i.e. a high tide in which the water rises to a certain level, recedes slightly, and then rises again. This phenomenon is mainly caused by non-linear effects like friction and self-advection (Zimmerman 1976).

Observations 1981-2010

A recent set of data, uploaded almost daily, was obtained from the database of the Dutch Seabird group (NZG/CVZ; currently managed by www.trektellen.nl). Seawatchers, a highly specialised and well trained group of amateur ornithologists, conducted systematic, year-round observations from vantage points along the Dutch coast (Camphuysen 1985, Camphuysen 2011). Although their main focus was the occurrence of true seabirds and migratory wildfowl and waders, they have always recorded marine mammals. These observations provide a long-term dataset from which long-term trends and seasonal patterns can be analysed (Camphuysen 2011). For this paper, we restrict our review to data collected at the seawatching site at Huisduinen (HD, figure 1) during 1981–2011. Harbour porpoises occur year-round in Dutch coastal waters, but with a distinct peak in January-March (Camphuysen 2011). For the Huisduinen data, only this period has been analysed. Seawatchers recorded the date, weather, and duration of the counts (start- and end-time), and reported sightings per hour of observation (n.hour^{-1}).

Observations in 2011

Standard seawatches from observation site Huisduinen (HD), following the exact same methods as in previous seasons, were continued to provide comparable data with earlier years. Observation teams were stationed at three additional locations: at Texel on the Wadden Sea dike near the NIOZ (ND), and near the southern tip of the island near a naval base (MB) (figure 1). In addition, observations were conducted from the dike at Den Helder

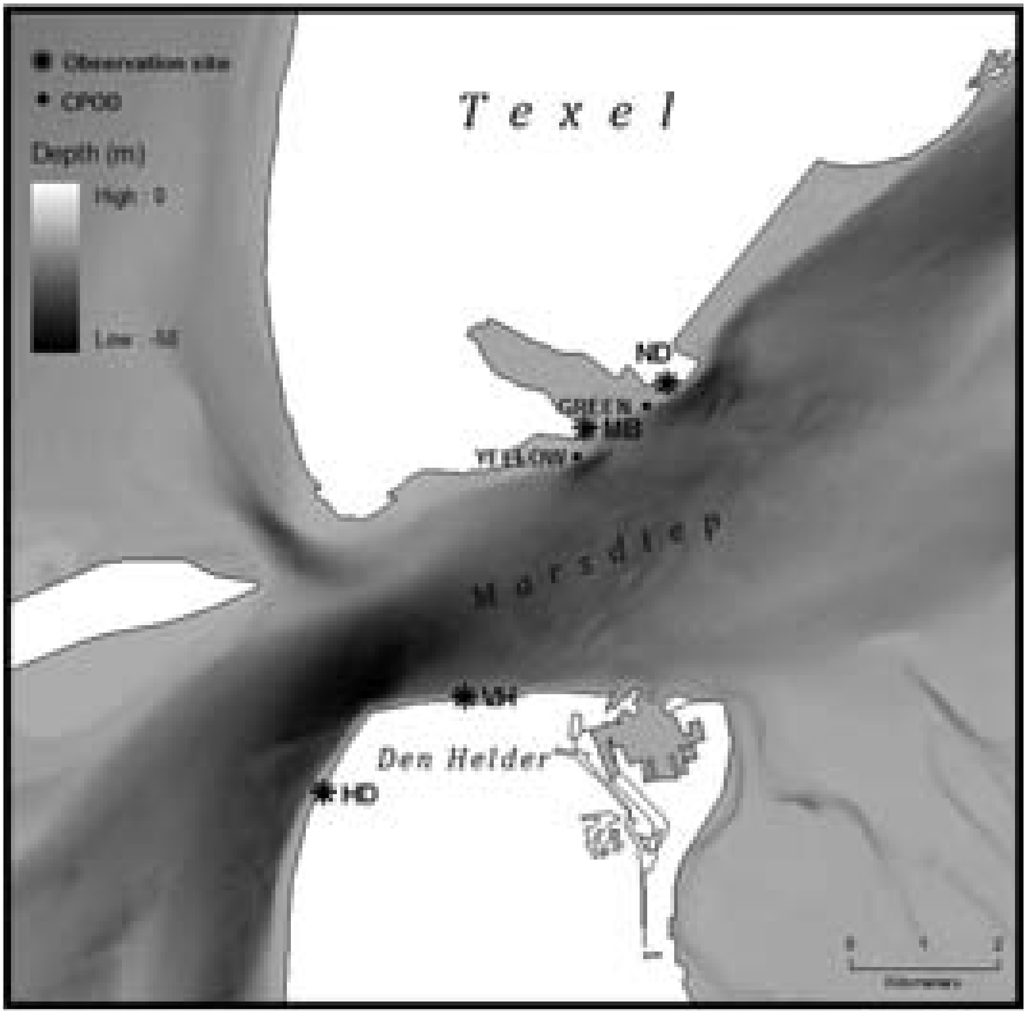

Figure 1. Map of the study area including bathymetry and locations of C-PODs (green and yellow). Standard seawatches were conducted from Huisduinen (HD). Further observation sites included the Wadden Sea dike near NIOZ at Texel (ND), the southern tip of Texel near a naval base (MB), and the dike at Den Helder ("Vlettenhelling"; VH). Green refers to the position of the first C-POD (#1482), attached to a green pole, yellow refers to the position of the second C-POD (#1481) attached to a yellow pole.

(Vlettenhelling; VH), some two km further east into the Wadden Sea relative to the Huisduinen seawatching site. Observations were made between 20 February and 20 April 2011 (516 hours of observation). However, data from April were excluded from the analysis because harbour porpoises were very scarce in that period (81 hours, 5 sightings). Observations were generally done by three observers, with one observer always at Huisduinen and the other two at any of the other observation sites. Effort was logged in 15 minute intervals, recording site, date, time, observer, meteorological conditions (wind direction and speed, cloud cover, visibility, precipitation), sea state, tide and any visual oceanographic features, such as fronts and tidal ripples. For each sighting, real time, distance and

Table 1. Observer effort (h) for visual observations in the Marsdiep area, February-April 2011. For the more detailed analysis of sightings, only the observer effort in bold was used: excluding data collected in April or when visibility was poor (see text); see figure 1 for site locations.

Location	Abbreviation	Feb	Mar	Apr	Total
Huisduinen	HD	**54**	**170**	34	258
Dike Den Helder	VH	**2**	**62**	15	79
NIOZ-dike Texel	ND	**10**	**61**	13	84
Naval base Texel	MB	**0**	**63**	19	82
Poor visibility		3	10	0	13
Total effort (sum)		69	366	81	516

angle were noted. To assess distance to the observer, 7x50 reticle binoculars were used. The reticles were converted to meters following Buckland et al. (1993).

Details about group composition, swimming direction and association with birds or oceanographic features were recorded for each sighting. Apart from harbour porpoises, sightings of seals (Phocidae; not identified to species) and vessels were recorded, under the assumption that high traffic densities and high seal abundances (or the absence of shipping and/or seals as the other extreme) could influence the behaviour and relative abundance of porpoises within the study area.

Weather permitting, six hours of observation per day were planned, with observer effort being equally distributed over the time of day and over the tide, to account for obvious factors that may affect the abundance and behaviour of harbour porpoises. During rough weather (high sea state), scheduled visual observations were cancelled. Data of observations conducted in poor visibility (less than 1000 m) were excluded from the analysis (effort and sightings; in total 13 h observation and six sightings; table 1). Data collected in April, when harbour porpoises appeared to be very scarce, were excluded from the analysis of factors associated with high abundances (a removal of 81 hours of observation, resulting into only five sightings). Information on tide and water level were obtained from the DONAR (Data Opslag Natte Rijkswaterstaatdatabase) made available by the Dutch Ministry of Infrastructure and the Environment at www.waterbase.nl. Time to high-tide was classified into four classes, with break points at high and low tide and exactly in between low and high tide. This was used to analyse the relation between porpoise abundance, behaviour and tide.

Passive acoustic monitoring

Because of their elusive nature, harbour porpoises are easily overlooked, particularly in windier conditions with rough seas. Therefore we used, in addition to our visual observations, passive acoustic monitoring devices, so-called Continuous Porpoise Detectors (hereafter called C-PODs) at two sites; one situated close to the NIOZ dike (53°00'N, 04°47'E) and the other one on the other side of the bay, close to the naval base (52°99'N, 04°77'E; figure 1). C-PODs record frequencies of 20 to 160 kHz and pick up harbour porpoise clicks. The click train filter developed by Tregenza (2011) was used to extract click trains from background noise. Porpoise click frequency is very high, between 100 and 160 kHz, which means that the sound propagation range is relatively small (Au 2000). Hence, the range of a C-POD is only around 250 meters horizontally, depending on

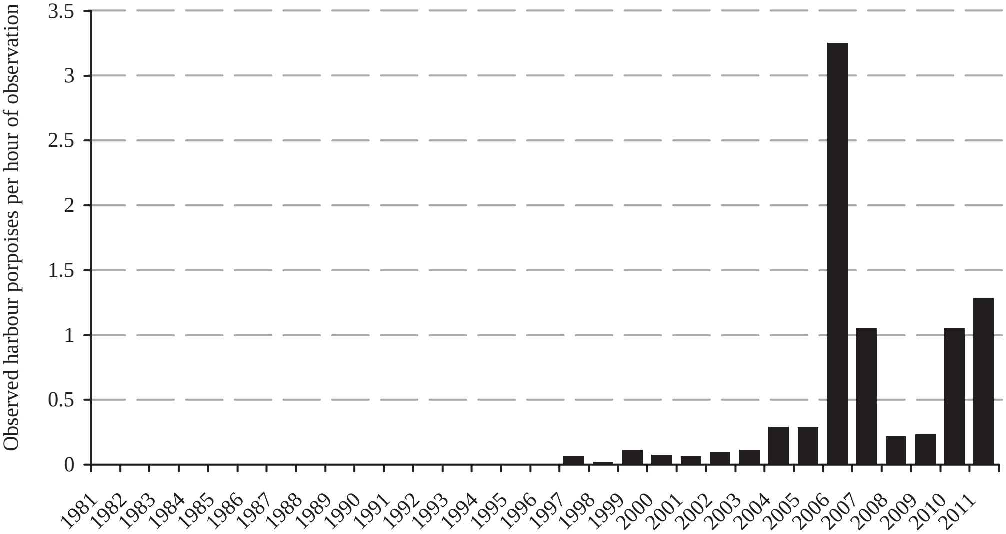

Figure 2. Harbour porpoises (*n*.h-1) in spring (January-March) at Huisduinen, based on systematic seawatching data, 1981-2011 (database NZG/Club van Zeetrekwaarnemers, www.trektellen.nl and unpublished annual reports Vogelwerkgroep Den Helder, 1996-2005).

sea conditions, but also on characteristics of the harbour porpoise clicks and orientation of the individual relative to the C-POD. The beam width of the sound emitted by porpoises is only about 12° (Au et al. 1999), hence a substantial number of porpoises in proximity of the C-POD may remain undetected.

To increase buoyancy, a buoy was glued to the C-POD. The C-POD was attached to an anchor with a 1 m long rope, so that the device was not in contact with the sea floor, in order to minimize the influence of sediment noise as much as possible. Once every few weeks, the SD memory cards of the C-PODs were replaced. The data on the memory cards was processed in the computer programme CPOD.exe, V2.012 (Tregenza 2011). From 8 February 2011 onwards, CPOD #1482 ("green pole") was deployed, and from the 17 February 2011 onwards a second CPOD (#1481, "yellow pole") was used (figure 1). Deployment depth during low tide was at approximately 9 and 14 m, respectively. Data recorded between 22 February and 7 April were used for comparison with our visual observations. In the software programme the trains of clicks were classified as low, moderate or high quality clicks on the basis of their probability to be a porpoise click (Tregenza 2011). In the analysis, only click trains of high and moderate quality were used. Low quality click trains were left out, because these were considered to include many false positive detections. The input high pass filter was kept at the normal setting (20 kHz) and the train filter was set on detecting only NBHF cetaceans (thus 'other cetacean' was not ticked). For the data analysis, the range was set at 117 to 155 kHz. The number of minutes per day during which porpoise clicks where detected (porpoise positive minutes – PPM) was used to describe the temporal trend in porpoise occurrence.

Results

Seawatching data Huisduinen 1981-2011

During systematic seawatching observations in Jan-Mar at Huisduinen between 1997 and 2011,

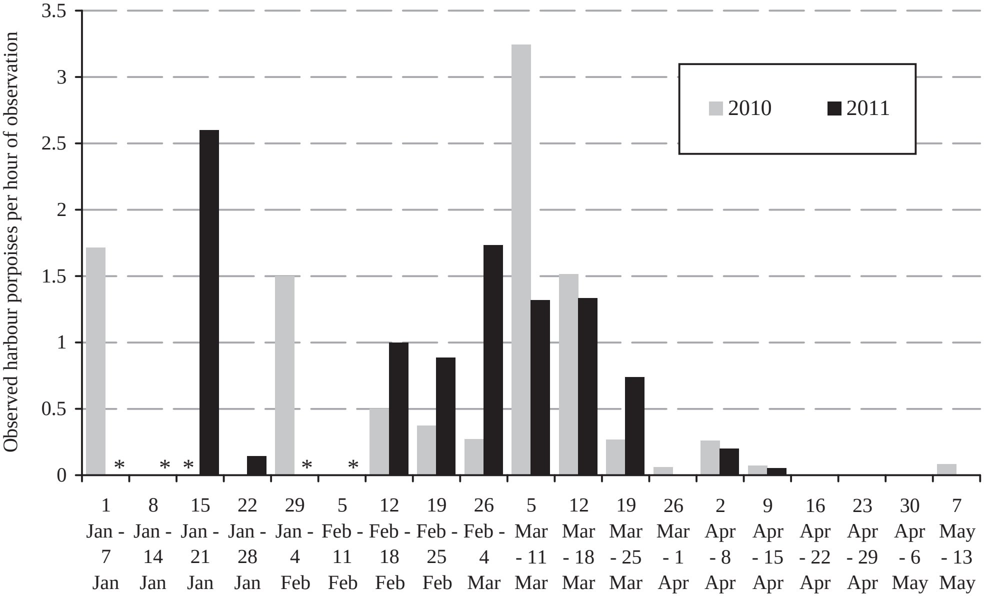

Figure 3. Weekly observed harbour porpoises (n.h^{-1}) at Huisduinen, January to mid-May, 2010 and 2011 (www.trektellen.nl). * no observation effort

1458 porpoises were recorded in 2400 hours of observation (mean 0.61 porpoises.h^{-1}); while none were seen and recorded in earlier years (1981-96; figure 2). The overall mean sighting frequencies in late winter did not exceed 0.20.h^{-1} until 2004. In 2006, sightings peaked at 3.25.h^{-1}, and from then on did not drop below 1.h^{-1}, with the exception of 2008 and 2009, when 0.25.h^{-1} were recorded. Spring 2011 was the season with the second most observations ever, with slightly higher numbers than in 2007 and 2010, but less than half the number of porpoises per hour as in 2006. In the first five months of both 2010 and 2011, sightings (n.h^{-1}) were most regular between late February and late March (figure 3). In both years, numbers declined markedly in April.

Marsdiep visual observations in 2011

Between 20 February and the end of March 2011, 605 harbour porpoises were recorded during more than 400 hours of observation (poor visibility conditions excluded; 1.53.h^{-1}). A first peak in sighting frequencies was recorded on 23 February, a second peak on 5-7 March, and a third period of relatively high numbers of sightings lasted from 12-30 March (figure 4). Very few animals were observed in April, after a marked decline in sighting frequencies during the last week of March. Harbour porpoises were most abundant off the naval base at Texel (MB, 2.4.h^{-1}), followed by the Wadden Sea dike at Texel (ND, 1.8.h^{-1}), seawatching site Huisduinen (HD, 1.3.h^{-1}), and the dike at Den Helder (VH, 0.75.h^{-1}). The mean pod size (±sd) was 1.31±0.59 per sighting (range=1-5).

Particular phases of the tidal cycle produced more sightings than others. Relatively high abundances were recorded around high tide (late flood and early ebb; figure 5). When a distinction is made between ebb and flood, higher abundances were found during flood. This pattern was found at most sites, but most clearly at the naval base. Along the dike of Den Helder near VH, however, a complete different pattern was found: here the highest abundance was observed directly after

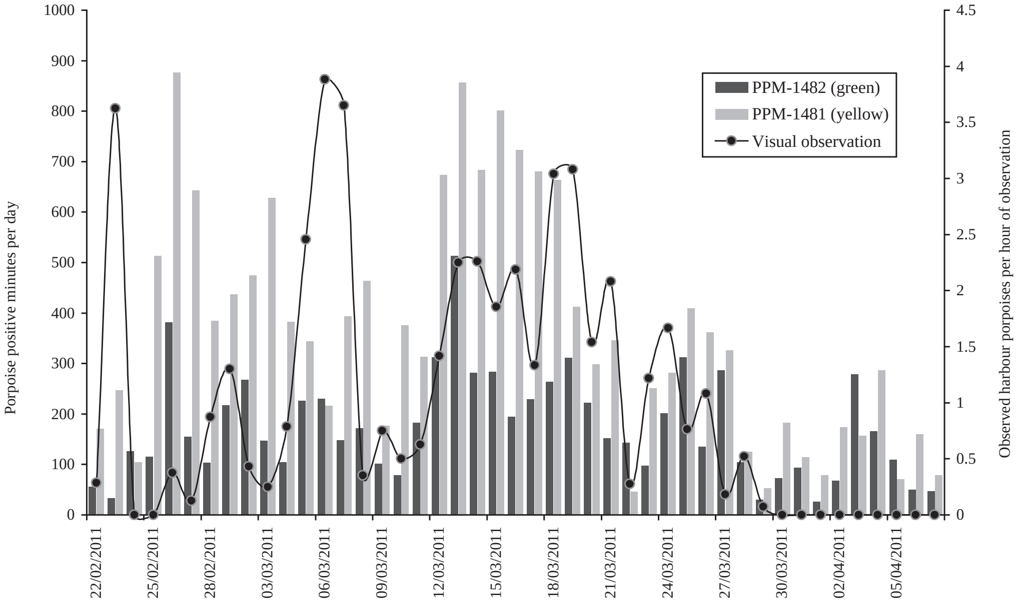

Figure 4. Acoustic detections of harbour porpoises (porpoise positive minutes per day; PPM) based on continuous recordings of two C-PODs in the Marsdiep area and visual detections during daytime (mean n.h^{-1}, indicated by ●) at all observation sites combined (animals per hour of observation). Estimates of porpoise sighting rates on days during which no observations were possible (bad weather or otherwise), are based on linear interpolations between the observation days.

low tide, during early flood, after which it decreased until the next early flood. The visual observations included impressions of the behaviour of porpoises. Some porpoises were accompanied with searching and plunge diving gulls, but by far most animals were seen away from the most prominent feeding frenzies of seabirds.

Acoustic observations in 2011

The number of porpoise positive minutes recorded per day by the C-PODs during the study period show a similar seasonal pattern as that obtained by visual observations (figure 4), but some disagreements in recordings (high peak in sightings with few acoustic recordings or vice versa) occurred on 23 February (many sightings), 26-27 February (frequent acoustic recordings) and around 6 March (many sightings). Daily activity indices derived from both C-PODs (using days that visual observations were conducted in the Marsdiep area) were positively correlated (Pearson's correlation based on log PPM per day: r^2= 0.67, df=43, t=5.9, P<0.00001). Also, the frequency of sightings and acoustic recordings for both C-PODs were positively correlated (mean log of PPM of C-POD 1481 ("yellow") per day versus log of daily sightings rate; r^2= 0.58, df=43, t=4.72, P=0.000025, mean log of PPM of C-POD 1482 ("green") per day versus log of daily sightings rate; r^2= 0.48, df=43, t=3.6, P=0.00079). Acoustic detections and visual detections were scarce in April.

Discussion

During this pilot study, a substantial data-set could be obtained and especially in March porpoises were observed on a regular basis, particularly near the coast of Texel. Although

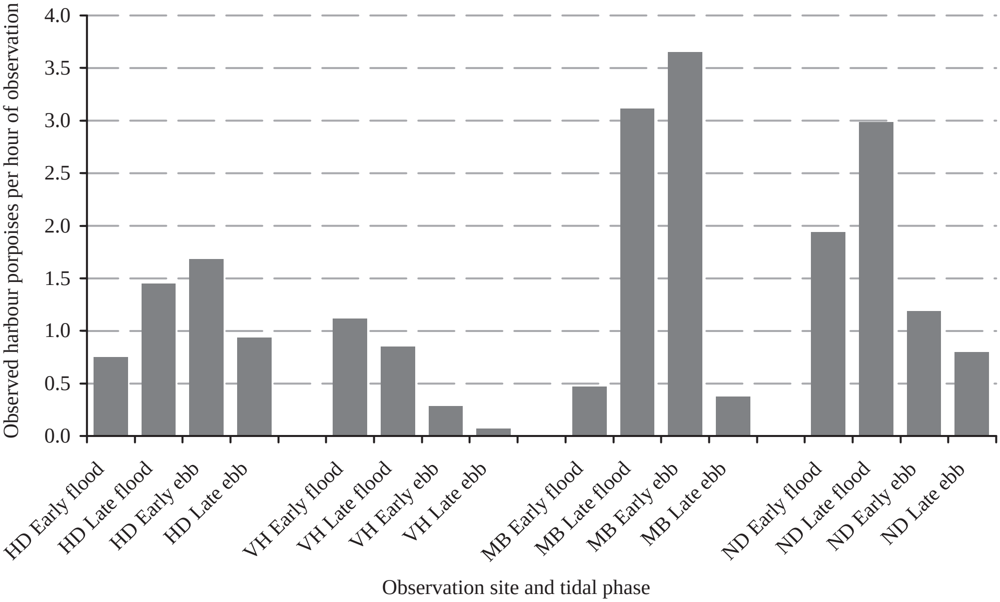

Figure 5. Visual detections of harbour porpoises ($n.h^{-1}$) at each observation site relative to the tidal phase.

at almost every location a relation between abundance and tide was found, the effect was most clear off the naval base (MB), where the abundance was highest around high tide. At the constant effort-site Huisduinen, the abundance of porpoises in spring 2011 ($1.53.h^{-1}$) was similar to 2007 and 2010, but higher than in 2008 and 2009 (mean $1.6.h^{-1}$ for the same observation period at Huisduinen, 2006-2010). From the perspective of our feasibility study, harbour porpoises were certainly numerous enough in early spring to have the Marsdiep serve as a study area for harbour porpoises in future years.

The higher sighting frequencies at the Texel side, particularly near the naval base, suggest that porpoises have a preference for this side of the Marsdiep. Alternatively, the more frequent sightings may have been caused by larger numbers of semi-resident animals (temporarily present on the spot; numerous re-sightings; movements within a particular area). Along the dike of Den Helder, more animals may have been just passing by and would have a lower chance to be recorded over and over again. Directed movements (all animals moving in the same direction) are common at Huisduinen, but on days with particularly high numbers of sightings, local movements (producing double-counts) prevailed.

Visual vs. acoustic observations

The abundance of harbour porpoises in the Marsdiep area during our study period was highly variable (figure 4), and the causes of fluctuations in numerical abundance are currently not well understood. The peak in sightings on 6-7 March consisted mainly of sightings at one observation site. The abrupt end of sightings in April was, like in other years, very distinct, and seems to point at a general departure towards deeper waters of the North Sea (cf. Camphuysen 2011). In contrast, recent coastal observations in 2013 suggest a high abundance of harbour porpoises in April (www.trektellen.nl).

Some peaks in sightings were not reflected in the acoustic data. Because acoustic recording were deployed during the same period in the same area, we were able to compare the acoustic recordings with indices of abundance recorded during visual observations. The acoustic data was expressed in porpoise positive minutes recorded per day. Porpoises use echolocation clicks to locate prey and to navigate, but when travelling, less clicks per minute are produced than when searching and hunting prey (Verfuβ et al. 2009). Hence, fluctuations in numbers of clicks does not necessarily represent variations in number of porpoises. However, PPM does not differentiate between variations in click frequency, and is therefore a more robust estimate of local occurrence. The relative strong and significant correlation between the visual sightings and C-POD recordings indicates that the acoustic recordings can indeed be used as a proxy of local porpoise abundance.

Both the acoustic and visual observations reveal a sudden disappearance in April. The historical seasonal pattern in abundance (Verwey 1975) was compared with the seasonality observed today (Camphuysen 2011). Verwey did not describe a sudden disappearance of porpoises in spring, nor a distinct peak season in late winter. In the old days, numbers gradually declined through May, June, and July. Interestingly, most recent observations in 2013 suggest a high abundance in April. This may be a single year event caused by cold weather conditions in March, or may be a first sign of a forward seasonal shift.

Competition and prey

The decline in numbers of porpoises in late winter described by Verwey (1975) may have been linked with the arrival of common bottlenose dolphins. Common bottlenose dolphins entered the Wadden Sea in early spring and in considerable numbers (ter Pelkwijk 1937, Viergever 1940), chasing the migrating Zuiderzee herring (*Clupea harengus*) (Verwey 1975). This annual event coincided with a seasonal decline in porpoise numbers. Bottlenose dolphins can be aggressive towards and even kill harbour porpoises (Ross & Wilson 1996), and their arrival could have been the main reason for porpoises to avoid the area during this period. The bottlenose dolphins have disappeared from the Marsdiep area and did so far not return (Camphuysen & Peet 2006). The abrupt decline in numbers in April as observed during this study can therefore not be explained by (aggressive) encounters with larger cetaceans.

According to Johnston et al. (2005), a strong relation with the distribution of their main prey species is an essential part of the harbour porpoise life-history. A link between the abundance of whiting (*Merlangius merlangus*) in the North Sea and the abundance of porpoises in late winter, as proposed by Verwey (1975), may still be valid today. Whiting, and also gobies (*Pomatoschistus* spp. - a common prey of immature porpoises in late winter), form the major part of adult harbour porpoise's diet in terms of mass and energy (M.F. Leopold, personal communication). The movements of harbour porpoises in this part of the North Sea can be related to the migration of some important prey species and a relation between the presence of porpoises in the Marsdiep area and the spring abundance of gobies has been suggested. The annual migration of gobies was described by Fonds (1973). The harbour porpoise is a relatively small, endothermic predator with limited energy storage capacity, dependent on foraging throughout the year without prolonged periods of fasting (Kastelein et al. 1997, Bjørge 2003). Future work in the Marsdiep area will need to focus at the foraging ecology of porpoises entering this tidal inlet to find a potential link with seasonal fluctuations of available prey.

Tide and foraging opportunities

The bathymetric composition of the site interacting with the tidal current could induce upwelling or relative high vorticity, enhancing particle transports and food availability in the water column (Mann & Lazier 1996). Such features could enhance the foraging opportunities for harbour porpoises (Jovallanos & Gaskin 1983, Johnston et al. 2005). If foraging opportunities off the naval basis did increase around high tide, movements of porpoises should be directed towards the area during flooding. Porpoises were in fact rarely seen to join these feeding frenzies in the turbulent areas around the shallows off the naval base. Their frequent occurrence in the area suggests other foraging opportunities that must receive more attention in future studies. Our results suggest that individual porpoises do not stay in the Marsdiep for long periods, but rather move in and out on a more or less daily basis.

Conclusions

In conclusion, the Marsdiep and its spring harbour porpoise population, provide excellent opportunities to study these elusive cetaceans in their natural habitat in the wild. During this pilot study, with relatively simple techniques, a considerable amount of new information on temporal and spatial aspects of the occurrence of porpoises in one of the most spectacular tidal inlets in the Wadden Sea was collected. The study opportunities off the naval basis at Texel are the most promising. The area is complex in its oceanography and bathymetry, and serves as a foraging area for several piscivorous top-predators. Advanced observation techniques may have to be deployed to pinpoint the exact positions of porpoises in that area. Such exact locations of sightings will then be combined with high resolution data on local bathymetry, currents and vorticity, water properties such as salinity, turbidity and temperature, and hopefully also with the availability of suitable prey, in order to achieve a better understanding of the ecology of harbour porpoises in this dynamic sea area.

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Samenvatting

Bruinvissen (*Phocoena phocoena*) in het Marsdiep: nieuw onderzoek in een historisch onderzoeksgebied

De bruinvis (*Phocoena phocoena*) keerde na een periode van afwezigheid van ongeveer drie decennia aan het eind van de vorige eeuw terug in de Nederlandse kustwateren. Bruinvissen zijn hier momenteel het meest aanwezig in de winter en het vroege voorjaar (december-maart). In het Marsdiep, de doorgang van de Noordzee naar de westelijke Waddenzee tussen Den Helder en het eiland Texel, werd de bruinvis in het midden van de 20ste eeuw bestudeerd. Dit onderzoek moest echter wor-

den gestopt wegens het verdwijnen van de bruinvis. Na de terugkeer van bruinvissen in de Nederlandse kustwateren is er in 2011 een haalbaarheidsonderzoek uitgevoerd om na te gaan of het opnieuw de moeite waard zou kunnen zijn om bruinvissen in het Marsdiep te bestuderen. De aantallen bruinvissen die in het voorjaar van 2011 werden gezien waren groot, maar weken niet wezenlijk af van wat er, op basis van zeetrektellingen bij Huisduinen, aan aantallen bruinvissen in de afgelopen vier seizoenen werd gezien. De meeste dieren werden waargenomen rond midden-maart en ze verdwenen in april. In vergelijking met het historische voorkomen lijkt deze piek in voorkomen twee tot drie maanden verschoven in de tijd. Waargenomen aantallen waren het hoogst aan de Texelse kant van het Marsdiep, vooral in de monding van de Mokbaai ter hoogte van de marinebasis op de zuidpunt van Texel. Op bijna alle observatieposten werden relatief veel waarnemingen gedaan rondom hoog water, bij de marinebasis was dat het meest uitgesproken. De waarnemingen suggereren dat, vooral tijdens hoogwater, deze plaats zuidelijk van de Mokbaai een voor bruinvissen aantrekkelijk gebied vormt, wellicht om te foerageren of om hydrodynamische redenen. De door Verwey waargenomen verplaatsing van bruinvissen "met het getijde mee" kon tijdens ons onderzoek niet worden bevestigd. Wij concluderen dat bij gelijkblijvende (of toenemende) aantallen bruinvissen in het gebied, het Marsdiep een uniek en zeer geschikt studiegebied zou kunnen zijn om bruinvissen ongestoord en in het wild te onderzoeken.

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