Effekter av omdirigering av sjöfart på alfågel och tumlare vid Hoburgs bank och Midsjöbankarna

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Underlagsrapport till havsplanering

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1 Sammanfattning

Midsjöbankarna och Hoburgs bank söder om Gotland är viktiga områden för de hotade populationerna av alfågel (*Clangula hyemalis*) och tumlare (*Phocoena phocoena*). Under det senaste årtiondet har en potentiell effekt av sjöfarten på dessa populationer, i närheten av bankarna, diskuterats. En möjlig omdirigering av sjöfarten har föreslagits, men tills vidare har inga åtgärder vidtagits. I Havs- och vattenmyndighetens rapport 2016:24 (Larsson 2016), beskrivs sjöfartens påverkan på alfågel och tumlare ur ett havsplaneringsperspektiv. Eftersom information gällande de flesta viktiga populationsparametrar har saknats för dessa ”svårtillgängliga” populationer så är denna tidigare utredning av kvalitativ och resonerande karaktär, utan kvantitativ analys av populationseffekter. Nyligen har fler användbara data framtagits gällande tumlaren i SAMBAH-projektet, vars mål var att beskriva utbredningsmönster och storlek av tumlarpopulationen. Gällande alfågeln utförde Nilsson (2016) nyligen en inventering på bankarna och en grundlig sammanställning av all tillgänglig kunskap gällande arten sammanställdes ytterligare i en alfågel ”action plan” 2015 (Hearn et al. 2015). Därtill har forskare vid Linnéuniversitetet nyligen sammanställt estimat av proportionen unga/vuxna honor under vintern, dvs. ett produktivitetsmått som kan användas för att beräkna årlig populationstillväxt. Även om informationen berörande dessa populationer fortfarande är begränsad så har vi nu en bättre möjlighet att kvantitativt beskriva alfågelns populationsdynamik och potentiella konflikt mellan sjöfart och viktiga tumlarhabitat. Målet med denna studie är att bygga vidare på den befintliga kunskapen och data, och kvantitativt beskriva inverkan av sjöfarten på dessa två viktiga och mycket skyddsvärda arter och undersöka vilken effekt en omdirigering av sjöfarten i intresseområdet potentiellt kan resultera i. Vi har strukturerat arbetet för att besvara fem specifika frågeställningar som ställts av Havs- och vattenmyndigheten:

1) Hur stor del av alfågelns populationsmortalitet kan tillskrivas sjöfarten i det aktuella området söder om Gotland?

2) Vad skulle det innebära för alfågelpopulationens status, hotnivå och framtidsutsikter om sjöfartens påverkan i det aktuella området uteblev?

3) På vilket sätt påverkas östersjötumlaren av sjöfarten i det aktuella området och vilka belägg finns för att sjöfarten medför en betydande skada på populationen?

4) Vore det ur tumlarens perspektiv fördelaktigt om sjöfarten inte rörde sig genom det aktuella området norr om Hoburgs bank utan istället färdades i djupvattenrutten, mellan Midsjöbankarna, där idag en del av trafiken går?

5) Är sjöfarten i området förenlig med det nya stora Natura 2000-område som nyligen beslutats om i syftet att skydda miljön kring Midsjöbankarna och vore det ur denna synpunkt riktigt att koncentrera sjötrafiken till ett stråk genom området (djupvattenrutten) för att minska skadan på Natura 2000-områdets värden?

För att besvara frågeställningar 1 och 2 gällande alfågeln har vi konstruerat en simpel deterministisk honbaserad åldersstrukturerad Leslie matrixmodell. För att kalibrera modellen har vi använt all tillgänglig information. Med hjälp av modellen har vi kunnat skapa en kontext för den befintliga kunskapen och därmed skapat oss en helhetsbild av populationsdynamiken. Adult överlevnad är den klart mest betydelsefulla parametern i modellen, och en förändring i denna parameter resulterar i en proportionellt stor förändring i populationstillväxten, i jämförelse med de andra parametrarna. Med hjälp av den estimerade populationsproduktiviteten och de befintliga populationsstorleksestimaten har vi kunnat estimera den årliga dödligheten.

Den årliga dödligheten har vi ytterligare kunnat dela upp i flera kategorier varav operationella oljeutsläpp utgör den väsentliga kategorin för denna studie. Endast en studie har bestämt storleksgraden av den årliga dödligheten pga. operationella oljeutsläpp (Larsson och Tydén 2005). Därutöver kan också ett index som beskriver antalet oljeskadade fåglar per år på Gotlands sydkust användas för att bilda en uppfattning av storleksordningen av dödligheten som kan tillskrivas sjöfarten (Larsson och Tydén 2005, Larsson 2016). Detta index tyder på att antalet oljeskadade fåglar har minskat de senaste åren. Vi har därför använt oss av två scenarier för att beskriva den relativa effekten av sjöfarten på alfågelpopulationen. I den ena har vi antagit att den årliga dödligheten pga. operationella oljeutsläpp i området omkring utsjöbankarna är 11 % (av antalet alfåglar omkring utsjöbankarna) medan det andra ”best case scenario” antar en dödlighet på 5 %. I resten av Östersjön antog vi att dödligheten pga. operationella oljeutsläpp konstant är 1 %. Om vi ser närmare på resultaten från ”the best case scenario” så kan dödligheten pga. sjöfarten i vårt intresseområde estimeras till över 1 % av den totala populationsstorleken årligen. Därför kan inverkan av sjöfarten i intresseområdet anses vara betydande. Om sjöfarten omdirigeras förväntar vi oss att dödligheten pga. oljeutsläpp minskar med 90 % (Forsman 2017). Detta skulle innebära, enligt modellen, att populationstillväxten blir positiv (Lambda stiger från 0.996 till 1.008) och populationen kommer att växa. Enligt modellresultaten kommer populationen att vara 12 % större år 2026 ifall sjöfarten omdirigeras i förhållande till en oförändrad sjöfart (eller 30 % större om en 11 % årlig dödlighet på grund av operationella oljeutsläpp antas).

För att besvara frågeställningarna 3-5 gällande tumlaren så har vi sammanställt den befintliga litteraturen gällande potentiella konflikter mellan tumlare och sjöfart. Eftersom ljud kan anses vara ett av de största hoten i området så har vi konstruerat en simpel ljudmodell. Vi har undersökt hur stor andel av viktiga tumlarhabitat som sammanfaller med de stora farlederna i intresseområdet. Av resultaten framkommer att skeppstrafiken sammanfaller med ca 1/3 av de viktiga tumlarhabitaten under sommarhalvåret (och även mer under vinterhalvåret). Sommarhalvåret kan antas vara viktigast med tanke på fortplantningen som sker under denna tid. På samma sätt har vi undersökt hur områden med för tumlaren relevanta ljudfrekvenser skapade av sjöfarten sammanfaller med de viktiga habitaten för tumlaren. Resultaten visar att 31-41 % av Natura 2000-området (som avser att skydda tumlare) sammanfaller med ljudnivåer som enligt litteraturen framkallar en flyktreaktion hos tumlare. Analysresultaten indikerar att en potentiell omdirigering av sjöfarten skulle innebära en klar minskning av områden som sammanfaller med viktiga tumlarhabitat, både gällande ljud och fysisk närvaro av skepp. Därför kan en eventuell omdirigering av skeppstrafik antas ha en positiv inverkan på populationen i intresseområdet och natura 2000-området, och därmed vara fördelaktigt ur tumlarens perspektiv, även om den slutliga inverkan på populationsdynamiken inte kan bestämmas kvantitativt.

2 Introduction

The offshore banks, Hoburg's bank and Midsjö banks south of Gotland are important areas for the declining and threatened species Long-tailed Duck (*Clangula hyemalis*) and harbour porpoise (*Phocoena phocoena*). During the last decade the potential indirect effect of shipping on these populations on the offshore banks has increasingly been discussed. A potential change in shipping routes with the aim to reduce the impact on these populations has been suggested. However, so far no measures to decrease the shipping intensity in the area have been taken. Potential effects of shipping intensity have qualitatively been described by Larsson (2016), however a more quantitative approach is lacking. This is most likely partly due to the fact that information regarding important population parameters on these "remote" species is scarce. New useful data has recently been collected and reported on harbour porpoises by the SAMBAH project aiming at estimating the size and distribution patterns of the Baltic population of harbour porpoises. New Long-tailed Duck surveys on the offshore banks were conducted in 2016 (Nilsson 2016). A thorough Long-tailed Duck action plan, summarizing all available knowledge on the species was compiled in 2015 (Hearn et al 2015), and most recently new values on productivity of the Long-tailed Duck population has been presented (Kjell Larsson submitted). Although still scarce there is now more data available than before. The main aim of this project is to build on these assessments already done and to quantitatively estimate the impact of shipping on the environment and particularly on these two "key" species utilizing the offshore banks. The work is structured to answer five specific questions asked by Havs- och vattenmyndigheten (SwAM: Swedish Agency for Marine and Water Management) (translated into English):

1) How large a proportion of the mortality in the Long-tailed Duck population is due to ship traffic in the area of interest south of Gotland?

2) What would the effect be on the Long-tailed Duck population in terms of status, level of threat and future prospective if shipping would not be allowed in the area?

3) In what way are harbour porpoises affected by shipping in the area of interest and what kind of evidence is available indicating a significant negative impact on the population?

4) Would it be beneficial for harbour porpoises if ships would not be passing the area of interest north of Hoburg's bank but instead in the Deep Water route in between the Midsjöbanks, where some of the ships are already passing?

5) Is the ship traffic in the area in agreement with the new enlarged Natura 2000 area with the purpose of protecting the environment around Midsjö

Banks and would it from this point of view be correct to concentrate the ship traffic to one route through the area (the Deep Water route) and thereby minimizing the negative impact on the values of the Nature 2000 area?

To be able to answer these questions all available information has been compiled and a matrix population model for the Long-tailed Duck has been built. A scenario was run in which shipping was relocated from the shipping lane north of the banks to the Deep Water route south of Hoburg's bank. Harbour porpoise distribution patterns have been assessed in relation to shipping patterns and underwater noise of relevant frequencies. These assessments have resulted in a more detailed picture of the potential impact of shipping on these vulnerable/threatened populations.

3 General study area and shipping

The study area is located in the central Baltic Sea with the area of interest being the offshore banks south of Gotland and surrounding areas (Figure 1). A thorough description of the areas and the environment is given by Larsson (2016) and will not be reproduced for this report. The shipping in the area is further described in the report by Forsman (2017) and the shipping will therefore not be described in detail in this report, however the same AIS data was used as described by Forsman (2017). The main shipping lanes are displayed in (Figure 1).

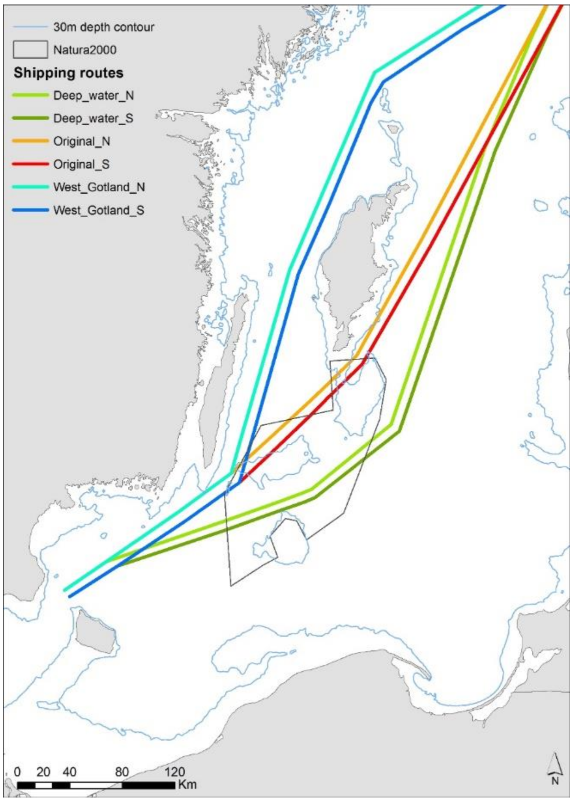

Figure 1. Study area, with current shipping routes depicted

4 Study species

4.1 Long-tailed Duck

The Long-tailed Duck is globally threatened and classified as vulnerable on the IUCN Red List. A recent and thorough literature and expert knowledge compilation on the Long-tailed Duck population was conducted for the work on the AEWA single species action plan (Hearn et al. 2015, see http://www.unep-aewa.org/en/document/draft-international-single-species-action-plan-conservation-long-tailed-duck-0). In that report a summary of the relevant information is given regarding population delineation and size estimate, demographic characteristics and population parameters and description of threats and mortality factors. All of these are essential components of the population model that has been built within this project with the aim to assess population trajectories and changes due to a change in shipping in the area of interest. Therefore, the current work relies heavily on the data presented in Hearn et al. (2015).

4.1.1 Population delineations and trends

The Long-tailed Duck is usually divided into four separate populations of which the West Siberian/North European population is of interest to this project (Hearn et al. 2015). The West Siberian/North European population breeds throughout Western Siberia with a smaller proportion also breeding in northern Finland, Sweden and Norway. It overwinters mainly in the Baltic Sea, with some birds overwintering along the mainland coast of the Barents Sea and the Norwegian Atlantic coast. As the main focus of this study is the Baltic Sea, the overwintering birds in the Baltic Sea have been defined as our study population (Figure 2). The birds overwintering in the Baltic are considered mainly to breed widespread in Western Siberia (Hearn et al. 2015). The Long-tailed Duck starts migrating from moulting areas in September, reaching the Eastern Baltic Sea from mid-September to October and areas further to the west between October and December. The spring migration starts in March and during late May the majority of the birds leave the Baltic Sea (Skov et al 2011). Around 25 % of the population (or probably even more during very cold winters) spend the winter on the Swedish offshore banks south of Gotland in the middle of the Baltic Sea. The size of the Baltic population has been estimated to about 1,500,000 birds (in 2007-2009), and has declined rapidly since the mid 1990's when the population was estimated to be around 4,272,000 (Skov et al. 2011). On the Swedish offshore banks the trend has been the same, declining from around 1,000,000 in 1992-1993 to 360,000 in 2007-2009. According to the most recent counts, the decline in recent years seems not to be as steep as up to 2009. Nilsson (2016) reported that the number of Long-tailed Ducks on the offshore banks in 2011 was 365,000 and 260,000 in 2016. The observed values fits well with a yearly population decline of 7 % until around 2009 and 2 % from 2011 onwards (Figure 3).

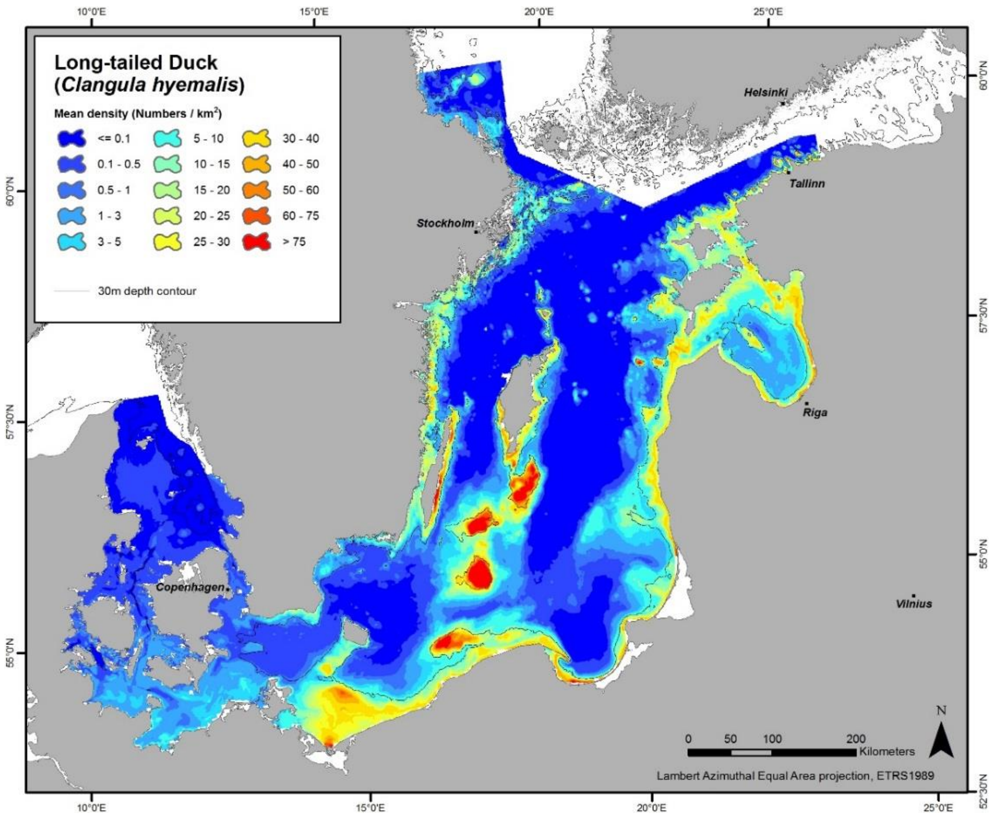

Figure 2. Non-breeding distribution of the long-tailed duck in the Baltic Sea from Skov et al. 2011.

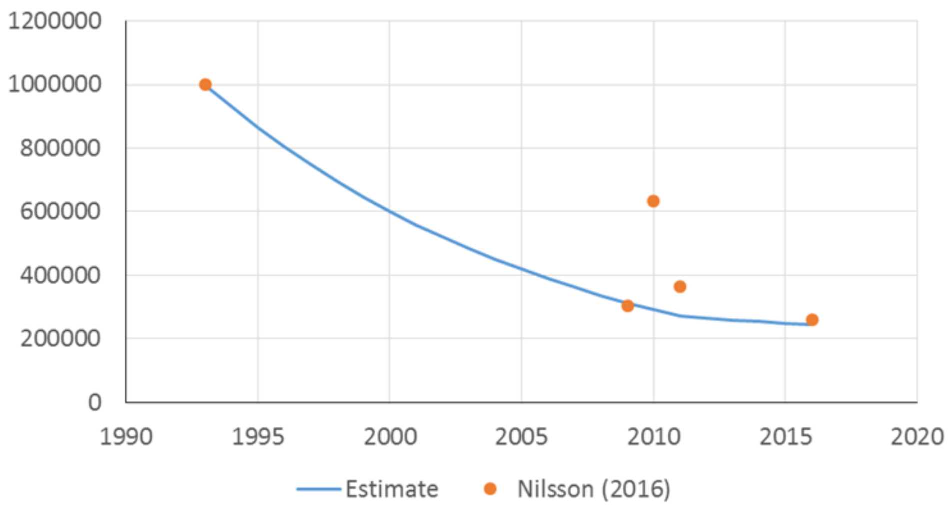

Figure 3 Counted birds on the Swedish offshore banks as reported by Nilsson (2016). The observed patterns fits a yearly population decline of 7 % until 2009 and about 2 % decline since 2011.

4.1.2 Demographics

Similar to other sea duck species, the Long-tailed Duck has a relatively high adult survival rate with a low production of young (Flint et al. 2015). However, there are relatively few studies on Long-tailed Duck demographics available (see e.g. Koneff et al. 2017, Hario et al. 2009, Schamber et al. 2009, Kjell Larsson submitted). Some examples listed below illustrate the large variability in demographic parameters obtained by different studies. An Islandic study indicated that the mean mortality rate of adults was 28 % (Cramp and Simmons 1977). An Alaskan study reported similar survival rates, 74 %, however they also indicated that the estimated survival rate is very low and might be applicable only locally (Schamber et al. 2009). A third study from Canada reported a survival rate of breeding females to be 85 % (Kellet and Alisauskas 2014) and the most recent study reports a survival rate of 81 % for the North American population (Koneff et al. 2017).

No specific survival rates have been suggested for the Baltic wintering population. However, the main mortality factors have been listed, and some values have been used. Hearn et al. (2015) for example judged that the combined mortality caused by recurrent operational oil discharges, fishing bycatch and hunting add up to 2-5 % yearly (on top of mortality due to other reasons). Larsson and Tydén (2005) indicated that more than 10 % are killed annually due to recurrent operational oil discharges on Hoburg's Bank, based on analyses of oiled / non-oiled birds that have drowned in fishing nets.

A study from the Baltic Sea indicated that the annual mean proportion of immature birds during 1996-2012 was on average 11.4 % (personal comment by Kjell Larsson in Hearn et al. 2015). While in more recent years (2008-2017) the mean proportion juveniles per adult female has been estimated to 22 % (Kjell Larsson submitted). These two proportions are not directly comparable as the ratio between adult females and males are not 50:50, but it still indicates an increase. The variation between years is large and Hario et al. (2009) reported a very low proportion of immature birds in 2006 for example, only 3 %, however based on a small sample size. The overall fecundity in North America as reported by Koneff et al. (2017) was estimated to 18 %. There are a few studies from North America and Iceland that have estimated fecundity parameters in more detail (Koneff et al. 2017, Schamber et al 2009 and Cramp and Simmons 1977). A range of parameters have been estimated including breeding propensity, clutch size, nesting success, duckling survival and juvenile survival. Flint (2015) developed a generic sea duck population model utilising generic population parameters. He motivates the model with the fact that in most seaduck populations there is a lack of information on several population parameters, and therefore a generic model can be useful for filling those gaps. This information helps us to fill knowledge gaps with realistic values regarding Long-tailed Duck population parameters as well.

In the study population in the Baltic Sea low recruitment together with a high adult mortality is thought to be the reason behind the observed population decline (Hario et al. 2009). There may be several drivers influencing the low productivity including altered habitat conditions and increased predation on the breeding grounds, potentially due to for example climate change. It can partly also be due to carry-over effects from the non-breeding grounds due to decreasing food resources (see Hearn et al. 2015 and references within). Other drivers relate to conditions in the non-breeding areas of which the most important is a decline in food resources, which can negatively affect breeding propensity due to lower fitness or direct mortality due to starvation. A reduction in food supply can potentially be due to reduced nutrient concentrations driven by improved eutrophication control in coastal areas, increasing water temperature and/or predator pressure on mussels from the invasive round goby (*Neogobius melanostomus*) with the mussels comprising 65-89 % of the diet of the round goby (Kornis et al. 2012). Diseases, toxins and vitamin deficiency are other mentioned causes. Anthropogenic constructions (e.g. offshore windfarms) in the non-breeding grounds is also a factors potentially contributing to the additive mortality (see Hearn et al. 2015).

The available quantitative demographic information available has been compiled and used to calibrate a population model, which reproduces the observed population trends, see methods section below (Chapter 5.1.2).

4.2 Harbour porpoise

The Baltic Proper population of harbour porpoises (*Phocoena phocoena*) is estimated to be around 500 animals and is considered critically endangered (SAMBAH 2016a). It is genetically (Wiemann et al. 2010), morphometrically (Galatius et al. 2012) and distribution-wise separated from the Belt Sea population (Sveegaard et al. 2012, SAMBAH 2016a). Bycatch in gillnet fisheries has been recognized as the primary threat for the survival of the Baltic harbour porpoise population (HELCOM 2013, Hammond et al. 2016). Contaminant levels (Berggren et al. 1999, Beineke et al. 2005) and underwater noise (ASCOBANS) are further contributing factors. Shipping is the main contributor of underwater noise and because harbour porpoises rely critically on sound for navigation, foraging and communication, increasing noise levels from shipping activities may have pronounced effect on behaviour, distribution and performance of this species.

4.2.1 Effect of the anthropogenic noise on harbour porpoises

Vessel noise is considered the dominant anthropogenic noise source in the world’s oceans at low frequencies (National Research Council 2003, Tyack 2008). These frequencies propagate with little loss to absorption and can therefore affect marine life over large ranges (Uricks 1983). Cetaceans are especially vulnerable to anthropogenic noise as they are critically dependent on sound to communicate, navigate, and in the case of toothed whales, to forage by echolocation.

The harbour porpoise echolocates around 110 to 150 kHz (Møhl and Andersen 1973) and has its most sensitive hearing between 80 and 140 kHz, with hearing thresholds below 1 kHz (Kastelein et al. 2002, 2010). The Marine Strategy

Framework Directive by the European Commission (European Commission 2010) considers 63 and 125 Hz as dominating frequencies of large ships in deep waters (Ross 1976, National Research Council 2003) and therefore as indicators for ambient noise pollution in marine habitats. Consequently, noise from large vessels within the above mentioned frequency bands is likely not audible to porpoises unless the received noise levels are very high above their hearing threshold for a given frequency. However, studies from the Baltic area show that large vessels can produce noise at the broad frequency range from 25 Hz up to 160 kHz (McKenna et al. 2012, Hermannsen et al. 2014) and therefore within important frequencies for porpoises. Additionally, porpoises frequently react to smaller vessels, with high-frequency components (0.25–63 kHz octave bands) of vessel noise significantly increase the probability of porpoise avoidance behaviour. However, most of the smaller vessels are not equipped with Automatic Identification System (AIS) and therefore there is no data available for this project about actual numbers and routes of such vessels.

Harbour porpoises have been shown to change behaviour in response to a range of different underwater noise sources, with most observable reactions being avoidance, masking and temporary threshold shift (TTS). Few studies, however, exists about reaction of porpoises to ship traffic-induced noise. The review by Tougaard et al. (2015) indicates that porpoises react to noise if it exceeds the hearing threshold by 30-50 dB. This was also experimentally tested by Dyndo et al (2015) where she showed that noise of 110 dB at 2000 Hz (and therefore 30 dB over hearing threshold) triggers porpoise behaviour.

Masking is another potential consequence of ship-induced noise and can be especially pronounced for porpoises as, compared to other cetaceans, they produce narrow band high frequency clicks which attenuate over short distances, and porpoises must, therefore, remain close to be able to communicate acoustically. Communication between mothers and calves can be especially vulnerable to masking. The maximum communication range between mother and calf was estimated to be around 500 m for porpoises (Clausen et al. 2010). However, as the porpoise communication occurs at 110 kHz and more (Clausen et al. 2010), any level of sound at 2 kHz and below is unlikely to affect mother-calf communication.

Temporary increase in the hearing threshold of porpoises induced by noise exposure (TTS) is another potential consequences of exposure to noise by porpoises, however lower frequencies (1-2 kHz) are less efficient in inducing TTS than higher, more audible for porpoises frequencies (Kastelein et al. 2014, see review by Tougaard et al. 2015) and, therefore, mostly sound levels over 170 dB of 2000 Hz are likely to cause TTS of harbour porpoises.

4.2.2 Effect of ship presence on harbour porpoises

It is difficult to distinguish between the effects of presence of big vessels on the behaviour of porpoises from the effect on the behaviour of noise made by the ships. Avoidance or changes in swimming direction at ranges up to 800–1000m has been reported (Dyndo et al. 2015, Palka and Hammond 2001).

5 Material and methods

5.1 Long-tailed Duck population model

The population model used for answering questions 1 and 2 is described below. Starting with a general description of the model structure. Continuing with listing and describing the population parameters used and defining the model population. Finally, the ship relocation scenario(s) used for assessing a potential population change due to possible relocation of the major shipping lane is described.

5.1.1 Model structure

A population model (also called population viability analysis, PVA) is useful for predicting the growth rate and trend of a population over time based on a set of population parameters. Mortality and productivity are the fundamental components of the model and the balance between these two are summarised in the statistical term lambda (λ).

Lambda (rate of population change), λ =1-mortality rate + recruitment rate.

Lambda describes the trajectory or growth rate of the population, and can be used for estimating future population size by simple multiplication (a lambda of 0.90 predicts a yearly 10 % population decrease, 1.10 a 10 % yearly increase, and 1.00 a stable population). By using a population model it is possible to interpret a specific life history parameter (a trait influencing reproduction or survival) relative to the other life history parameters. This can be very useful in management for assessing the relative importance of a proposed management actions. In this study the key questions were how mortality, indirectly due to shipping, is related to Long-tailed Duck population dynamics, and how a change in shipping patterns could affect the population dynamics in the future.

There are different demographic modelling methods available and matrix modelling is a commonly used approach (Frederiksen et al. 2014). For this project a basic female-based deterministic age-structure Leslie matrix model was developed (Flint 2015). A similar model was constructed by Christensen and Hounisen (2014) aiming to assess impact of hunting restrictions on the Common Eider (*Somateria mollissima*) population in the Baltic Sea. The Long-tailed Duck PopulationModel (LPM) was based on the generic seaduck model by Flint (2015). Because there is an excess of males in the population of Long-tailed Ducks and the fertility of males is zero, it is sufficient to only consider females in the model.

The LPM model is defined as a pre-breeding census model with three age classes where juvenile survival is modelled as a fecundity (productivity) parameter (S0) and therefore the fecundity value per age class is the same as the number of juveniles per female that joins age class 1 (Figure 4). Fecundity per age classes is the product of breeding propensity (proportion breeding birds), breeding season survival, clutch size, sex ratio, nesting success, duckling survival and juvenile survival. The fecundity changes with age class. Survival is defined as 1-mortality. Estimates on mortality were retrieved from literature as well as through calibration, i.e. calibrating the model to fit observed population trends and productivity values. The survival changes with age class, similar to fecundity. The fecundity and survival values are inserted into a matrix, which further allows us to estimate future population sizes, and changes in population trajectories caused by changes in any of the demographic parameters. The model was constructed using the R package "popbio" (Stuben and Milligan 2007).

Because there is an indication of increased fecundity and potentially also increased survival in recent years from around 2012, the model was divided into three "submodels". The first submodel (M1) describes the decline until around 2012, whereafter the second submodel (M2) is describing a less steep decline as indicated by the most recent surveys, and assuming the same mortality due to recurrent operational oil discharges throughout the study years. Finally the third submodel (M3) assumes an increased survival due to a decrease in oiling mortality in the study area in addition to the increased fecundity described by M2.

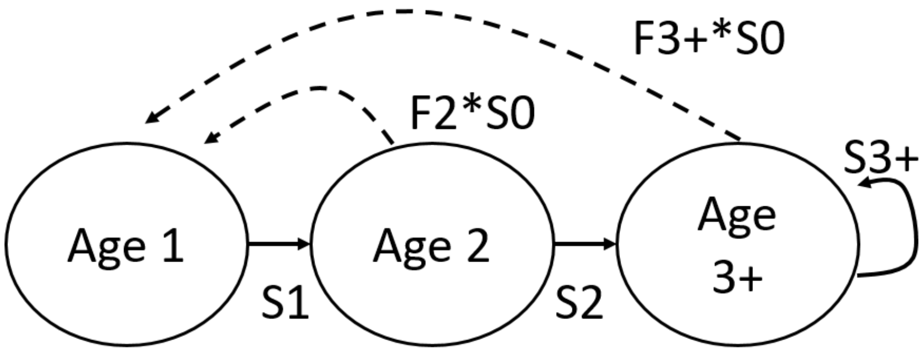

Figure 4. Flowchart of the age structured matrix model. S stand for survival and F for fecundity, because the model is a pre-breeding census model the survival of Age class 0 (which is never counted) is inserted as a fecundity parameter, modified from Flint (2015).

5.1.2 Population model parameters

Data on population parameters to fit the model was obtained from literature (Table 1). Many of the lower level fecundity values, which are multiplied to yield the total fecundity (productivity) per age class, are obtained from the North American studies. However, the values have been chosen, or calibrated, to fit the observed mean fecundity in the Baltic population. By also including lower level fecundity values (and not only the overall value) a better understanding of the observed productivity is achieved which is based on number of juveniles per adult female observed (female age ratio) on the wintering ground and analysed based on photographing bird flocks (Kjell Larsson submitted). For model M2 and M3, we used the mean female age ratio at a stopover site in Finland in spring during 2012-2017 (Kjell Larsson submitted). We corrected the value for male bias which gives us a value of 0.21. This value is higher than the overall mean reported for 2008-2017 which is 0.22 and when corrected for male bias 0.165. We have two reasons for using this higher fecundity value 1) because there seem to be a regional difference in the distribution of immature birds (Kjell Larsson submitted), and we assume this bias is less when the birds are migrating to the breeding grounds and 2) because this value fits well with the observed changes in the population sizes (Figure 3).

The mortality rates, used for calculating survival rates, were divided into 4 mortality factors (Table 1). The three most important mortality factors were according to Hearn et al. 2015 recurrent operational oil discharges, fishing bycatch and hunting, the fourth class includes all other potential factors combined. The size of "the other factors" was obtained by calibration against the other values in the model. On the three main factors there is at least a judgement on the level of magnitude of mortality, which is not available for the "other factors". By dividing the overall mortality into different groups, it was possible to further only manipulate the mortality caused by shipping (or more specifically oiling), and thereby assess the potential effect on the population after a hypothetical rerouting of ships.

Illegal oil spills have been monitored during most of the study period and there is a clear relationship with the major shipping lane passing north of the Swedish offshore banks (HELCOM 2016, Figure 5). However, most of the recurrent operational oil discharges most likely go unnoticed and there is no clear correlation between number of oiled birds found on Gotland and number of observed oil spills (Larsson 2016, Larsson and Tydén 2005, Larsson and Tydén 2011). Information on mortality due to recurrent operational oil discharges comes from two sources: 1) proportion of oiled birds drowned in fishing nets on Hoburg's Bank 2000-2004 (Larsson and Tydén 2005) and 2) from an index based on beached oiled birds counted on southern Gotland starting in the year 1996 (Larsson and Tydén 2005, Larsson and Tydén 2011, Larsson 2016,). The proportion of oiled birds in fishing nets could be regarded as the least unbiased estimate of actual proportion of oiled birds and was used in the population model. However, according to the beach bird index there has been a potential decrease in number of oiled birds (Larsson 2016, Figure 6). This decline is larger than would be expected based on only accounting for a decrease in population size. This might potentially be due to fewer oil spills (see Larsson 2016). Adding to this, there might be an increase in scavenging from foxes, eagles or other predators, which might induce variations of the index. However, to account for this potential decrease in oiling mortality it was assumed an arbitrary potential decrease in recurrent operational oil discharges from 11 to 5 % on the offshore banks and ran this as an alternative scenario.

Table 1. The parameters used in the population model. M1= submodel describing population trend until 2012, M2= submodel describing population trend from 2012 onwards assuming no change in oiling mortality, M3= submodel 3 describing population trend from 2012 onwards assuming a 50 % decrease in oiling mortality. A description of the parameter is included together with a reference.

	Parameter	M1	M2	M3	Description	References
Fecundity	Breeding propensity age 1	0	as M1	as M1	1 year old birds do not breed	Flint 2015
	Breeding propensity Age 2	0.28	as M1	as M1	Proportion of 2 year old birds breeding	Koneff et al. 2017
	Breeding propensity Age 3+	0.88	as M1	as M1	Proportion of 3+ year old birds breeding	Koneff et al. 2017
	Breeding seasonal survival	0.99	as M1	as M1	Breeding female mortality	Flint 2015
	Clutch size	7.05	as M1	as M1	Number of eggs	Koneff et al. 2017, (Schamber et al. 2009, reported 7.1)
	Sex ratio	0.5	as M1	as M1	Male:female ratio	Flint 2015
	Nesting success Age 2	0.21	0.256	as M2	Proportion successfully hatched, lower nesting success for first time breeders in accordance with Flint 2015	
	Nesting success Age 3+	0.38	0.46	as M2	proportion successfully hatched	M1: mean of Koneff et al. 2017 and Schamber et al. 2009 M2: Koneff et al. 2017
	Duckling survival	0.17	0.24	as M2	proportion successfully fledged M1 increased slightly to fit 0.11 overall	M1: mean of Koneff et al. 2017 and Schamber et al. 2009 M2: Koneff et al. 2017
	Juvenile survival	0.63	as M1	as M1	proportion juveniles surviving to the next age class	Koneff et al. 2016
	Fecundity Age 1	0	As M1	as M1	Multiplying all individual fertility parameters	
	Fecundity Age 2	0.02	0.04	as M2	Multiplying all individual fertility parameters	
	Fecundity Age 3+	0.12	0.21	as M2	Multiplying all individual fertility and Larsson (submitted)	M1: Larsson quoted in Hearn 2015, M2: Larsson (submitted)

	Parameter	M1	M2	M3	Description	References
Mortality	Recurrent operational oil spills	0.035	as M1	0.02	0.11 on the Swedish offshore Banks, based on proportion in fishing nets.	M1: Larsson and Tydén 2005, M3: assumed based on Larsson 2016
	Fishing bycatch	0.02	as M1	as M1	Assumed to be 2 %, based on literature	Zydelis et al. 2009, Bellebaum et al. 2013, Hearn et al. 2015
	Hunting	0.01	as M1	as M1	Based on hunting statistics	Hearn et al. 2015.
	Other mortality	0.095	as M1	as M1	"Estimated/assumed" to fit trend	Reviewed in Hearn et al. 2015
Survival	**Sub-adult survival**	0.74	as M1	0.755	10 % less than adult survival	same ratio to adult as in Koneff et al. 2017
	Adult survival (1-mortaility)	0.84	as M1	0.855		Based on the mortality prop. listed

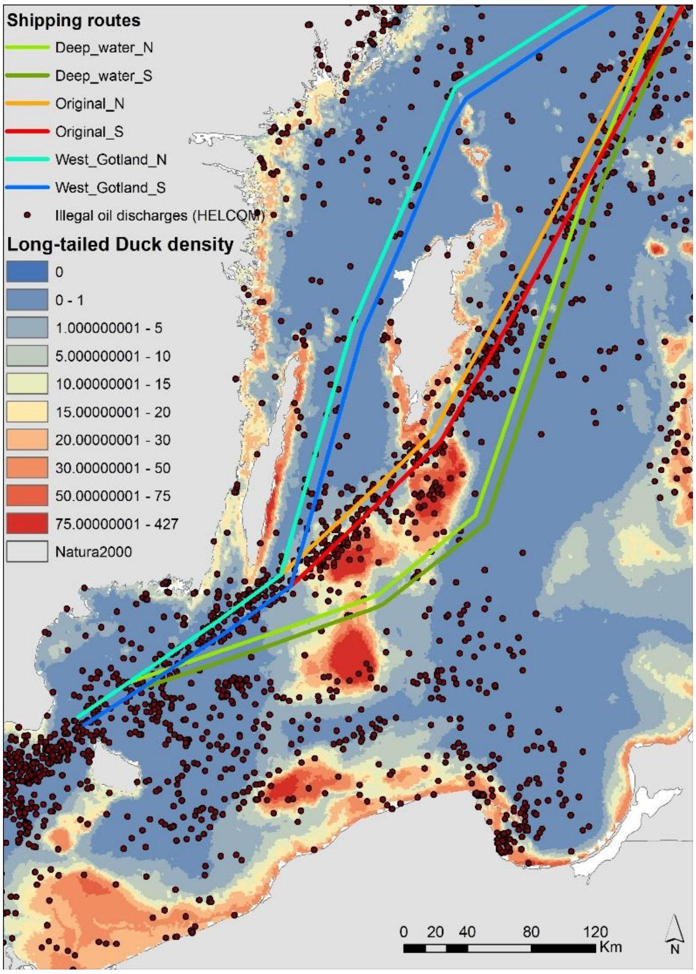

Figure 5. Visualization of Long-tailed Duck densities, main shipping lanes and observed illegal oil spills 1998-2015 obtained from the HELCOM data portal (maps.HELCOM.fi).

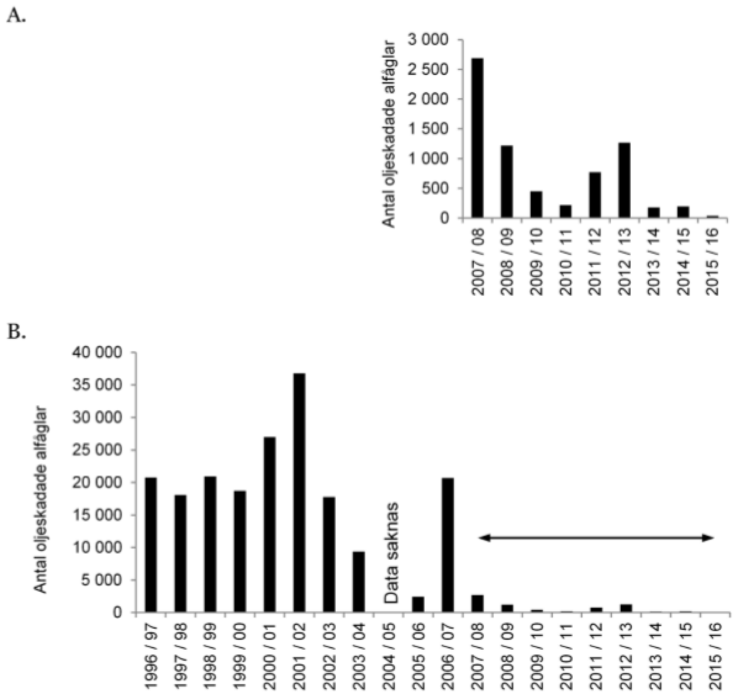

Figure 6. Index of weekly observed number of oiled Long-tailed Ducks on southern Gotland, summed per year. The upper chart A. is zooming in on the years between 2007 and 2016, while B is showing all years. The figure is copied from Larsson 2016.

5.1.3 Model population, size and projection

The LPM model population is defined as Long-tailed Duck females wintering in the Baltic Sea. The model period starts in 1992/1993 when the population was estimated to be 4,272,000 (Durinck et al 1994). The proportion females in the population was set to 0.43 (Hario et al. 2009, Larsson submitted), which equates to a female population size of 1,836,960. The age structured population sizes were set to 220,435 1 year old, 185,166 2-year-old and 1,431,359 3 years or older birds. The population model was projected on the years spanning from 1993-2026.

5.1.4 Elasticity, proportional importance of parameters

Elasticity is used for describing the relative change in recruitment rate (or lambda) when the demographic parameters in the models are changed. The estimated elasticity values should be interpreted as the increase in lambda in % when the specific parameter is changed by 1 %. For example, if the elasticity value of adult female survival is 0.50 then every 1 % change in adult survival would result in a 0.5 % increase in lambda (Flint 2015). Elasticity is therefore highly useful for assessing the importance of each parameter in the model.

5.1.5 Future scenario – Rerouting of shipping

A rerouting of shipping was "simulated" in another project (Forsman 2017). Ships were moved from the current main shipping lane crossing the study areas north of Hoburg's bank to the shipping lane south of the banks. The same study further assessed how recurrent operational oil discharges, in relation to probability of exposure to birds, would change if the ship traffic in this "middle lane" would be redirected to the southern Deep Water lane. As a result of the "simulation" in the Forsman (2017) the authors report a 10 times lower exposure rate of oiling to birds, taking into account the "dispersion time" and wind direction (see Forsman 2017). It was therefore assumed that this reduction would reduce oiling mortality by 10 times on the offshore bank population. As a result, it was assumed that after a rerouting of ship traffic, the oiling mortality in submodel M2 would decrease in the study area from 11 % to 1.1 % and in submodel M3 from 5 % to 0.5 %. This would result in a total population mortality due to oiling to be around 1 % and 0.9 % respectively in submodels M2 and M3. The changes in lambda and proportional change in bird numbers were assessed after 10 years to illustrate the effect of rerouting the ship traffic.

5.2 Harbour porpoise assessment

5.2.1 Harbour porpoise distribution

During summer season, the highest density of porpoises is found on and around the offshore banks south of Gotland and east of Öland: around the Hoburg's and Northern and Southern Midsjö Banks in the Baltic Proper especially in May – August, during the reproduction period (SAMBAH 2016a). These areas are considered essential and probably the main breeding area for the Baltic Proper harbour porpoise population (ASCOBANS 2016). During the winter season, especially during January – March, porpoises are more dispersed (SAMBAH 2016a).

The southern border of the study area is based on the spatial separation between the Belt Sea and Baltic harbour porpoise populations during May – October according to SAMBAH (2016b). The northern border is based on the spatial extent of the SAMBAH project (see Figure 7).

Habitat suitability classes were defined based on porpoise detection probabilities estimated during the SAMBAH project in summer (May – October) and winter (November – April) (SAMBAH 2016a). The study area was divided into 5 non-nested habitat suitability classes based on quantiles of detection probability: 10, 20, 40, 60 and 80 % (Figure 7). 10 % shows areas with highest detection probability and, therefore, most suitable areas for porpoises corresponding well with areas of high importance for porpoises indicated by the SAMBAH project (see Figure 10 of SAMBAH 2016a).

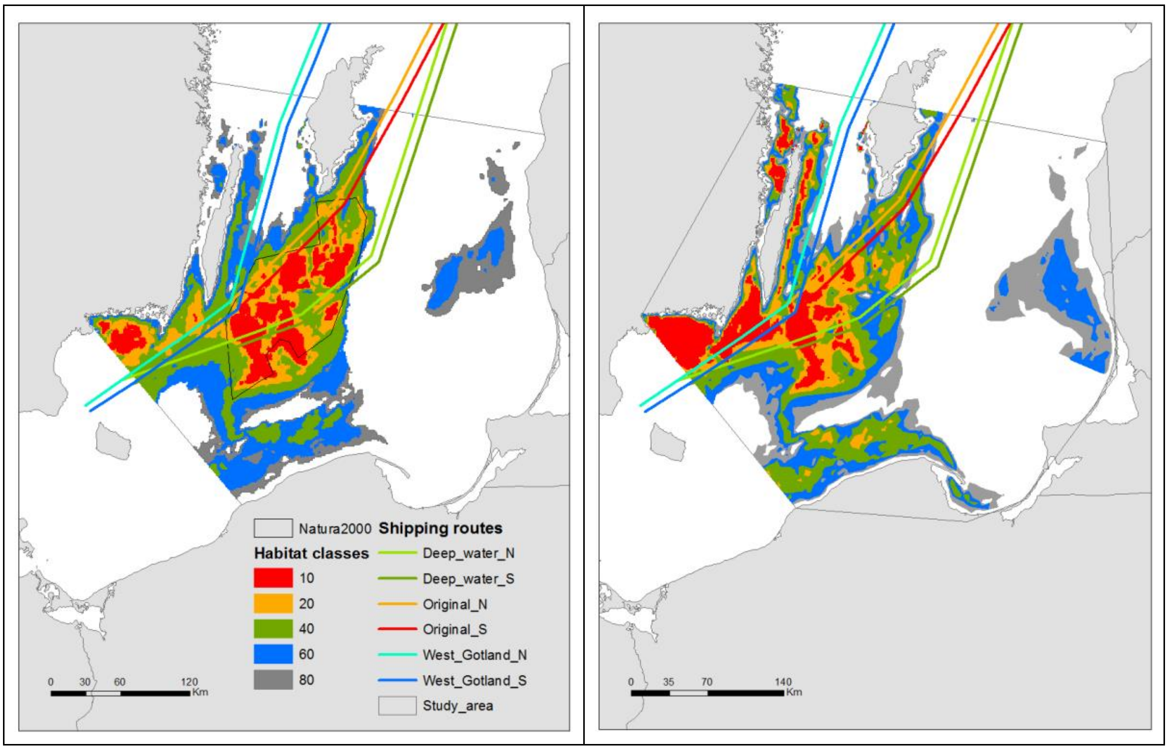

Figure 7. Summer (May-October) (left) and winter (November – April) (right) porpoise habitat classes estimated in SAMBAH project (SAMBAH 2016a). The graph depicts also the main shipping routes and Natura 2000 protected area with the study area.

5.2.2 Effect of the current anthropogenic ambient noise on porpoises

HELCOM, within the EU LIFE Baltic Sea Information on the Acoustic Soundscape (BIAS) project, has produced anthropogenic noise maps for three frequencies: 63, 125 and 2000 Hz as indicators for ambient noise pollution in marine habitats (BIAS LIFE11 ENV/SE 841, www.bias-project.eu, HELCOM 2014). As porpoises are not likely to react to the first two frequencies, all parameters used in the current analysis were based on 2000 Hz. The hearing threshold for porpoises at 2000 Hz is approximately 80 dB re 1µPa (rms) (hereafter referred to as dB) (Kastelein et al. 2002). In order to assess the potential effect of ambient noise produced by large vessels on Baltic harbour porpoise population, the areas important to porpoises and the Natura 2000 area were overlapped with 2000 Hz sound level at 110 dB (inducing avoidance) and 170 dB (causing TTS) (Dyndo et al. 2015, see chapter 4.2.1).

For the purpose of this project, a simplified model was applied to assess the effect of noise on porpoises based on basic sound propagation equations and time-integrated AIS ship traffic with a generalised sound source for all ships. All analyses were confined to the third octave with 2 kHz centre frequency. Cumulative noise map over the course of 6 months for summer (April - September) and winter (October - March) were derived based on existing shipping AIS data for three vessel categories (passenger, cargo and tankers) to represent the existing and relocated shipping routes. The shipping density was expressed as the number of ships that passed through a specific grid cell within the respective 6 months, i.e. the applied shipping density provides a time-integrated magnitude. Noise levels were therefore also estimated as cumulative magnitudes for the full 6 months period and then transferred to sound pressure levels that correspond to rms sound pressure levels (SPL).

The above approach is significantly different from the computationally more demanding approach of the BIAS project in the Baltic Sea (Helcom 2014, Nikolopoulos et al., 2016, www.bias-project.eu) The BIAS approach used AIS snapshots, so narrowband source spectra depending on ship speed and size could be associated with individual ships. Sound propagation was then numerically modelled for each snapshot, so that a large number of noise maps could be generated and statistical analysis carried out at each gridpoint. The BIAS approach therefore permits the calculation of percentiles and median noise levels. The approach applied in this study is in contrast rather dose based, i.e. the received noise dose over 6 months is calculated at each gridpoint and then converted to rms levels. As the noise evaluation methods had to be comparable for shipping traffic before and after rerouting the BIAS results were not used for this purpose.

5.2.2.1 Terms and definitions

In particular the following sound level definitions were applied:

Sound pressure level: $\mathrm{SPL} = 10 \log \frac{1/T \int_0^T p(t)^2 \mathrm{d}t}{p_0^2} = 10 \log \left(\frac{p_{\mathrm{rms}}}{p_0}\right)^2$

Sound exposure level: $\mathrm{SEL} = 10 \log \frac{\int_0^T p(t)^2 \mathrm{d}t}{p_0^2\, T_0} = \mathrm{SPL} + 10 \log (T)$

Both levels are relative to $p_0 = 1\ \mu\mathrm{Pa}$, integration over time t is carried out for the $T = 180$ days and referenced with $T_0 = 1$ s for the SEL. The instantaneous sound pressure is expressed by p, the rms sound pressure by p_{rms}. The difference between SEL and SPL is 72 dB with the SEL as the dose-like parameter being the higher one. For assumptions and equations that are true for both sound levels, SL will be used in the following.

For conversion between third octave and 1 Hz bandwidth data:

$$\mathrm{SPL}_{1/3} = \mathrm{SPL}_{f_\mathrm{m}} + 10 \log (f_2 - f_1)$$

where $1/3$ indicates the third octave and f_m, f_1, and f_2 are the center frequency, lower and upper frequency limit of the third octave, respectively. For the 2 kHz third octave to be analysed in the following, these values are f_m = 199f Hz, f_1=1778 Hz, and f_2=2239 Hz, so the difference between $\mathrm{SPL}_{1/3}$ and $\mathrm{SPL}_{f_\mathrm{m}}$ is 26.6 dB for the 2 kHz third octave.

5.2.2.2 Source levels

A common source level in the 2 kHz third octave band was applied for all ships. In the BIAS project numerical modelling applied narrow band source spectra for individual ships depending on speed and size. For a typical speed of 10 knots and a maximum length of 400 m, they applied narrowband levels below 130 dB re 1 µPa @ 1m / √Hz (Folgelot et al. 2016). The underlying model (Wales and Heitmeyer 2002) yields a narrowband level of 125.6 dB re 1 µPa @ 1m / √Hz, if no correction for ship speed and size is applied. Conversion to third octave levels yield 150 – 160 dB re 1 µPa @ 1m for the 2 kHz third octave. A more recent evaluation of noise levels radiated from commercial ships in the Santa Barbara Channel off the US West Coast showed third octave level between 160 and 170 dB re 1 µPa in the 1 kHz third octave (McKenna et al. 2012). For none of the analysed ship types the third octave level (@ 1 kHz) was below 160 dB re 1 µPa @ 1m. It appears therefore reasonable to apply the top noise levels derived from the narrowband spectra: The source level of the individual ship in the 2 kHz third octave band was set to 160 dB re 1 µPa @ 1m.

The total source level for each grid cell was determined by integration over time. For a typical ship speed of 11 knots (Forsman 2017) and a cell length of 250 m, it takes the ship about Δt= "1 min" to pass the cell, where the diagonal path through the cell has already been taken into account. For N ship passages through a grid cell, the total time of noise generation is therefore

$$T_{\mathrm{ship}} = N\,\Delta t$$

and the time integral in the above SL equations become

$$\int_0^T p(t)^2 \mathrm{d}t = p_{\mathrm{ship}}^2 T_{\mathrm{ship}}$$

where p_{ship} corresponds to the pressure associated with the single ship source sound level of 160 dB.

5.2.2.3 Transmission loss

The production of noise maps requires to simulate the noise at each individual grid point, as it is received from all the different sources. In this simplified analysis, the transmission loss TL between each source and receiver point is determined by a geometrical approach that is based on the cylindrical and spherical spreading laws:

$$\mathrm{TL} = 20 \log R \text{ (Spherical spreading law)}$$

$$\mathrm{TL} = 10 \log R \text{ (Cylindrical spreading law)}$$

A realistic geometrical spreading is typically of the form

$$\mathrm{TL} = A \log R$$

with A being a constant between 10 for perfectly cylindrical spreading and 20 for perfectly spherical spreading. Here, a standard value of $A = 15$ is applied. A minimum distance of $A = 10$ m is applied, if source and receiver cell are identical.

The received sound level RL for the above transmission losses is described by

$$\mathrm{RL} = \mathrm{SL} - \mathrm{TL}$$

The transmission loss between each source and receiver location is calculated individually. All received levels at a grid point are then combined into the total received level by summing the squared pressures.

5.2.3 Effect of current ship presence on harbour porpoises

AIS data for three vessel categories (passenger, cargo and tankers) were summed over two seasons: summer 2015 (April – September) and winter 2015 (October – March) and were used as indication of traffic intensity in the study area. 'High-traffic' areas were defined as 10 % quantiles, similarly to estimation of habitat suitability classes. A 1 km buffer was added around high-traffic areas to indicate potential areas of impact (Dyndo et al. 2015, Palka and Hammond 2001). The percentage of area of each habitat class and Natura 2000 area overlapping with areas of high-traffic including the buffer was then calculated.

5.2.4 Effect of anthropogenic ambient noise of reallocated traffic on harbour porpoises

In order to reallocate all traffic from Original to Deep Water shipping route (Figure 7), only traffic level classified as 'High-traffic' (Figure 8, Figure 16) was shifted as occasional traffic won't have a pronounced effect on porpoise behaviour and distribution. Further, occasional traffic occurs at less defined routes and is therefore difficult to relocate unless a given area is completely closed for traffic and this is not the case for the Natura 2000 area. The study area was first divided into polygons (Figure 8) and the mean value of AIS was calculated for each cell in each polygon of "Original Line" and added to each cell of "Deep Line" corresponding to Original Lines' polygon. Traffic intensity along the Original line was then set to zero. It was assumed that the traffic intensity between north and south direction of the same line was the same. Two scenarios regarding traffic relocation were run: one assuming no change in width of the new shipping line based on the assumption that the Deep Water route in its current width is able to sustain the increase in traffic (Forsman 2017) and a second scenario where the new width was calculated proportional to traffic intensity.
A relationship was established between traffic intensity and the width of shipping route based on original AIS data from 2015 (along current shipping lines) (Figure 9) and calculated new width of the Deep Water shipping line with new traffic intensity. This was done by calculating width-current traffic intensity relationship along three equally spaced transects along each of the three current lines and fitting a simple linear model to the relationships defined by these nine points. Separate analyses were run for summer and winter. The

new traffic intensity was then measured along the Deep Water line at the same transect locations using the linear equation to calculate the corresponding width.

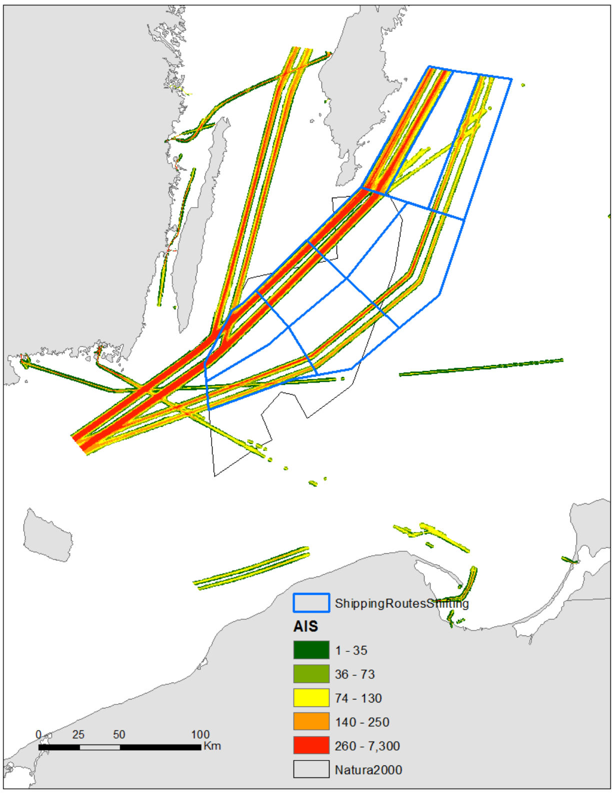

Figure 8. High-traffic line divided into polygons used to reshuffle traffic from Original and Deep Water traffic line.

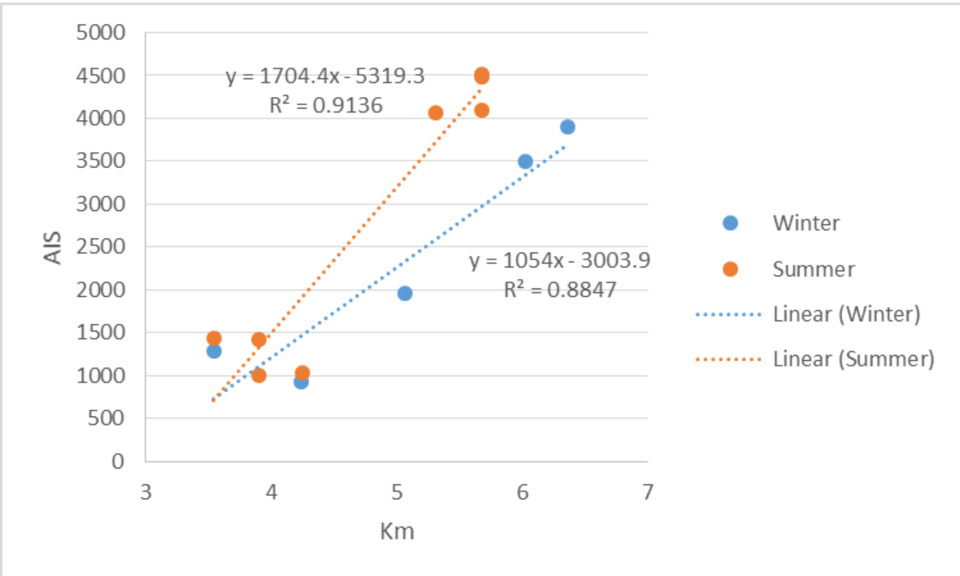

Figure 9. Relationship between traffic intensity and width of shipping routes for winter and summer season.

The AIS distribution for the new width was estimated based on the assumption that most traffic would be along the current Deep Water line and the remaining traffic distributed along the new width (Figure 10). Accordingly, 50 % of the traffic was allocated along the width of the current Deep Water line (depicted as orange colours in Figure 10) and the rest distributed over the new width (depicted by green colour in Figure 10). This proportion was based on the current traffic allocation along existing "Original Line", the shipping lane passing north of the offshore banks.

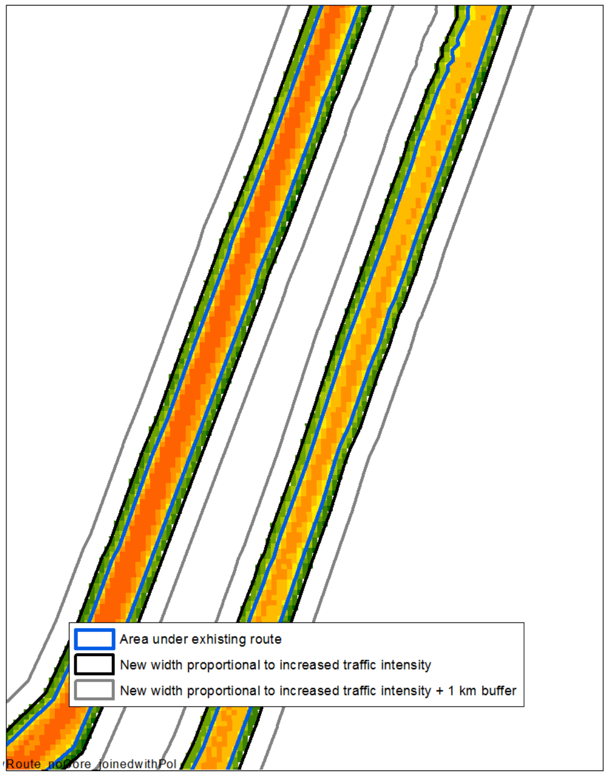

Figure 10. Schematic representation of the three area types used in the analysis: area under existing Deep Water route, area of the new width along the relocated traffic of Deep Water line and the same area with 1 km buffer.

5.2.5 Effect of reallocated ship presence on harbour porpoises

Similar to the analysis of the impact of noise, two scenarios for the relocated line were run: with and without width change. A 1 km buffer was added to the area used by high-traffic of the relocated route of both these two scenarios as indication of impact area as described in the introduction. Then the spatial overlap between habitat classes, Natura 2000 area and the relocated shipping routes was calculated for the two scenarios. The area and position of the West Gotland route was not changed for any of the analyses and was also included in the "area overlap" analysis.

6 Results

6.1 Question 1 - Long-tailed Duck mortality related to shipping

The constructed LPM model creates a context for evaluating the previously reported potential mortality due to recurrent operational oil discharges, estimated to be 11.8 % based on oiled birds found in fishing nets on Hoburg's bank (Larsson and Tydén 2005). This scenario was tested (11 % mortality on the banks = submodel M2), as well as an assumed reduced oiling mortality of 5 % on the offshore banks (M3). In other areas we assumed an average constant oiling mortality of 1 %. In Table 2 the yearly population sizes predicted by the models are shown together with the number of birds potentially killed by oiling on the Swedish offshore banks. The result indicates that if the oiling mortality on the banks would be 11 % yearly then around 2.75 % of the total population would be killed annually due to oiling mortality. If the oiling mortality on the banks on the other hand has been reduce to around 5 %, then the proportion of the population killed would be around 1.2 % of the total population.

Table 2. Model prediction, population size and annual oiling mortality on the total population and on the "Swedish offshore Bank population", for submodels M2 and M3.

Year	Pop. size M2	Pop. size M3	Oiling mort. M2	Oiling mort. M3	Mort. on banks M2	Mort. on banks M3
1993	1,836,960		64,294		50,516	
1994	1,696,469		59,376		46,653	
1995	1,573,696		55,079		43,277	
1996	1,461,198		51,142		40,183	
1997	1,356,149		47,465		37,294	
1998	1,258,612		44,051		34,612	
1999	1,168,133		40,885		32,124	
2000	1,084,158		37,946		29,814	
2001	1,006,216		35,218		27,671	
2002	933,878		32,686		25,682	
2003	866,741		30,336		23,835	
2004	804,430		28,155		22,122	
2005	746,599		26,131		20,531	
2006	692,925		24,252		19,055	
2007	643,110		22,509		17,686	
2008	596,877		20,891		16,414	
2009	553,967		19,389		15,234	
2010	514,141		17,995		14,139	

Year	Pop. size M2	Pop. size M3	Oiling mort. M2	Oiling mort. M3	Mort. on banks M2	Mort. on banks M3
2011	477,179		16,701		13,122	
2012	478,418	485,575	16,745	9,712	13,156	6,070
2013	470,326	484,995	16,461	9,700	12,934	6,062
2014	459,659	481,701	16,088	9,634	12,641	6,021
2015	450,591	479,788	15,771	9,596	12,391	5,997
2016	441,802	77,989	15,463	9,560	12,150	5,975

How realistic are these predictions? In other words, how realistic is our LPM model? To answer this question, it is necessary to examine the model results. The constructed matrix model has three parts or submodels. The first (M1) describes the steep decline from 1992/1993 until around 2012, with a lambda of 0.93 or an annual growth rate of -7 % (
Table 3). The second submodel (M2) describes the more moderate decline observed since 2012, with a lambda of 0.98 or growth rate of -2 % (
Table 3) assuming the same oiling mortality. The third submodel, the same as model M2 but further also assuming a decline in oiling mortality on the banks from 11 % to 5 % resulted in a lambda of 0.996, indicating an almost stable population with an annual growth rate of -0.4 % (
Table 3). The model matrixes are shown in
Table 4, Table 5 and Table 6.

When plotting the population trajectories predicted by the models, they fit rather well with the observed numbers during the two Baltic wide surveys, conducted in 1992-1993 and in 2007-2009 (Figure 11, Figure 12). This is not a surprise, since the model has been calibrated to fit the observation. However, the values also fit the observed productivity (Kjell Larsson personal comment in Hearn et al. 2015) during the two periods, before and after ~2012, ~0.12 and ~0.22 respectively. The modelled population trajectory also fits the population estimates on the banks made by Nilsson (2016) in 2016, assuming the offshore bank population is 25 % of the total population (Table 2). Further, the observed productivity fits the lower level fecundity parameters (or a combination of these) reported from different studies in North America (Table 1). As all these pieces of information combined produces the observed trend it gives some confidence in that the LPM model describes the demographic patterns driving the Baltic Long-tailed Duck population in general terms. Even if there is a very large uncertainty around the specific values due to a lack of real data. One could assume that if the estimated fecundity is reasonable then the survival should also be reasonable otherwise the model would not be able to reproduce the population trend observed assuming the population estimates are approximately correct. To indicate that there is a lot of variability the matrix models were also fitted with upper and lower standard errors around mean productivity as measured in winter and spring by Kjell Larsson (submitted). The upper and lower population trajectories are plotted in Figure 11 and Figure 12. It is however important to note that there is also variability around survival, however this is not included in the predictions.

How much of the mortality should be allocated to the different factors? This question is challenging to answer due to incomplete evidence of other factors than local oil pollution on the offshore banks and lack of data on the level of connectivity between the Long-tailed Duck populations on the banks and the populations in coastal areas. However, the values used for oiling are based on the only quantitative estimate on oiled birds available (Larsson and Tydén 2005). This is also backed up by the number of oiled birds found on the coast of Gotland, reported by Larsson and Tydén (2005) and Larsson (2016). The proportion of mortality due to oiling, hunting and bycatch used in the model is also of a similar order of magnitude as the range suggested by Hearn et al. (2015), 2-5 %. However, there is a lot of uncertainty, as already indicated, coupled to each of these values and therefor the results should only be regarded as indicative, a summary of the knowledge obtained so far.

It is not possible to estimate a realistic total level of uncertainty because of the lack of required data for making such estimates. It is, however possible to compare the relative contribution of the different parameters in the model, by looking at the elasticity values (
Table 4, Figure 14, Figure 15), and based on that get a better understanding of how the lambda changes when one of the parameter changes. Or, from another point of view, how much an error in one of the parameters affects the results. Based on the elasticity values we can conclude that a change in adult survival has clearly the largest effect on lambda or growth rate (
Table 4, Figure 14, Figure 15). Therefore, only a small change in survival has a larger effect on the population trajectory than a corresponding change in fecundity. On the other hand, an error in the estimation of the survival rate has also a larger influence on the estimated growth rate in comparison with the fecundity rate.

Only the mortality of oiling as a result of shipping has been reported on here. However, there might also be an indirect effect of shipping intensity on mortality due to displacement from suitable habitats. This could have a population effect due to birds forced to use less favourable feeding habitats, causing potentially starvation mortality or reduced fecundity due to for example reduced breeding propensity. It has not been possible to estimate this potential population effect in detail, however it was assumed to be relatively small in the current situation (<1 %), when the population is small and space can be assumed not to be a limiting factor. Nevertheless, this might mean that the effect of shipping might be slightly higher than estimated. Or, on the other hand if the estimate of oiling is too high it might be compensated by a small level of mortality due to habitat displacement, which is otherwise not accounted for. Therefore, a current mortality of around 1 % due to shipping in the area of interest is judged as realistic, and because the additive effect on survival, and the relative importance of survival on the population trajectory the effect cannot be considered to be negligible. In the chapter below the potential effect of re-routing ships is further reported on.

Table 3. Estimated growth rate (Lambda) and predicted female population size in 2026, for each submodel.

Model	Lambda	Estimated pop. size in 2026
M1: population parameters 1993-2012	0.928	155,836
M2: 2012->, increased fecund.	0.980	361,644
M3: 2012->, increased fecund. + lower oiling mortality	0.996	458,972
M2 + rerouted shipping	1.006	470,523
M3 + rerouted shipping	1.008	514,784

Table 4. Model matrix for the submodel M1. Fecundity (Fec.) and survival (Sur.) is indicated for each age class. The elasticity results is shown for both the fecundity and survival parameters.

Fec. Age 1 = 0 Elasticity = 0	Fec. Age 2 = 0.02 Elasticity = 0.0014	Fec. Age 3 = 0.12 Elasticity = 0.080
Sur. Age 1->2 =0.74 Elasticity = 0.081	0	0
0	Sur. Age 2->3 = 0.84 Elasticity = 0.080	Sur. Age >3 = 0.84 Elasticity = 0.759

Table 5. Model matrix for the submodel M2. Fecundity (Fec.) and survival (Sur.) is indicated for each age class. The elasticity results are shown for both the fecundity and survival parameters.

Fec. Age 1 = 0 Elasticity = 0	Fec. Age 2 = 0.04 Elasticity = 0.004	Fec. Age 3 = 0.21 Elasticity = 0.11
Sur. Age 1->2 = 0.74 Elasticity = 0.114	0	0

0	Sur. Age 2->3 = 0.84 Elasticity = 0.110	Sur. Age >3 = 0.84 Elasticity = 0.662

Table 6. Model matrix for the submodel M3. Fecundity (Fec.) and survival (Sur.) is indicated for each age class. The elasticity results are shown for both the fecundity and survival parameters.

Fec. Age 1 = 0 Elasticity = 0	Fec. Age 2 = 0.04 Elasticity = 0.003	Fec. Age 3 = 0.21 Elasticity = 0.110
Sur. Age 1->2 = 0.755 Elasticity = 0.113	0	0
0	Sur. Age 2->3 = 0.855 Elasticity = 0.110	Sur. Age >3 = 0.855 Elasticity = 0.664

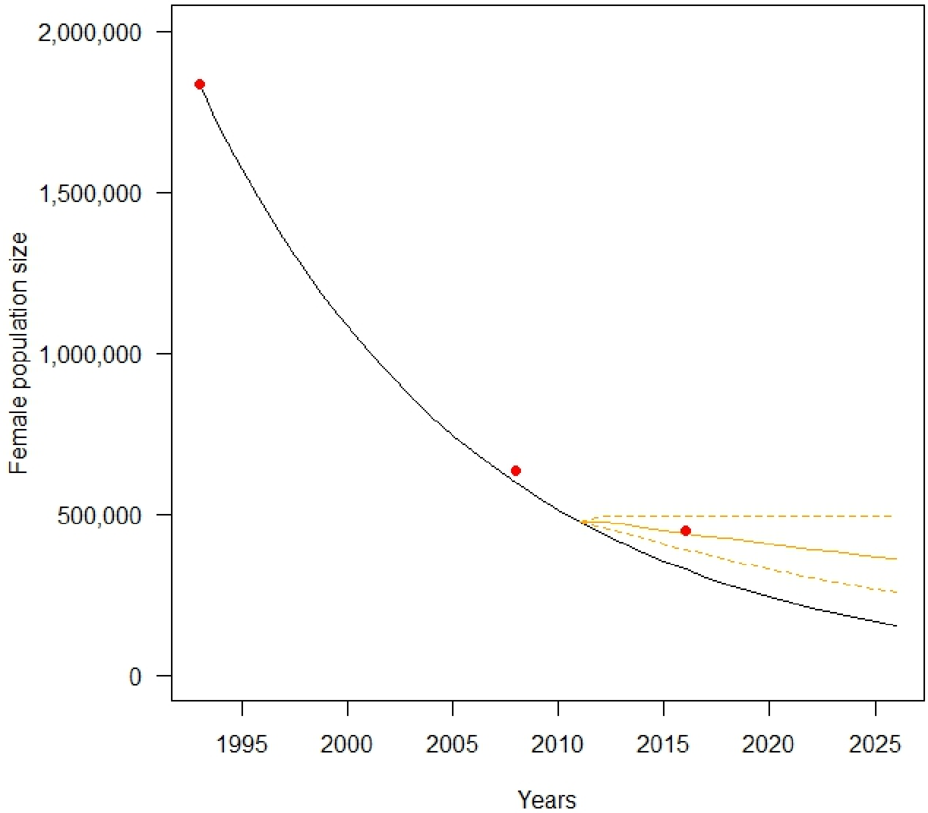

Figure 11. Population trajectories predicted by the LPM model. The black line indicates the trajectory of submodel M1 with a lambda of 0.93. The orange line indicates the population trajectory predicted according to submodel M2 with a lambda of 0.98. The dashed lines

indicate the model fitted with either upper or lower standard error around the mean productivity rate (fecundity) estimated in winter and spring in Larsson (submitted). The red dots indicate observed numbers during surveys conducted in 1993-1994, in 2007-2009 and in 2016. The 2016 number is extrapolated to the total female population based on assuming the Swedish offshore bank population being 25 % of the total population.

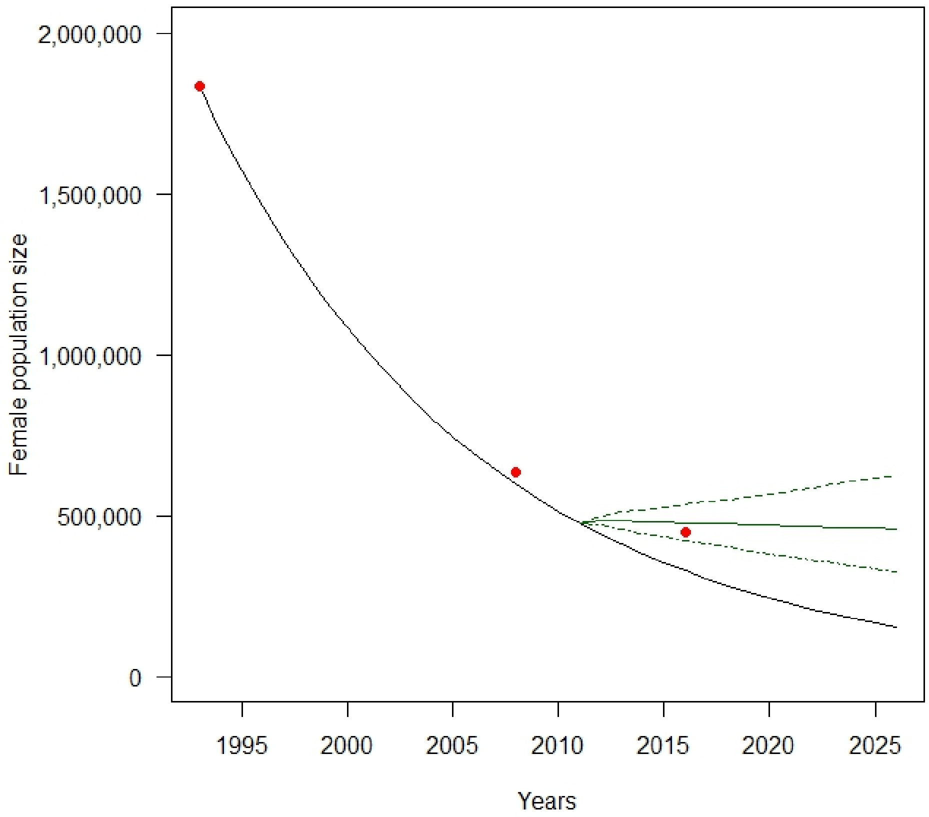

Figure 12. Population trajectories predicted by the population models. The black line indicates the trajectory of submodel M1 with a lambda of 0.93. The green line indicates the population trajectory predicted according to submodel M3 with a lambda of 0.996. The dashed lines indicate the model fitted with either upper or lower standard error around the mean productivity rate (fecundity) estimated in winter and spring in Larsson (submitted). The red dots indicate observed numbers during surveys conducted in 1993-1994, in 2007-2009 and in 2016. The 2016 number is extrapolated to the total female population based on assuming the Swedish offshore bank population being 25 % of the total population.

6.2 Question 2 - Long-tailed Duck population level effect in the case without shipping?

Assuming a 10 time less exposure risk for birds to oiling when ships are rerouted to the Deep Water lane south of the banks would mean that the 11 % mortality due to oiling on the offshore banks would be reduced to 1.1 % or on average 1 % of the whole population. This would mean an increase in lambda to >1 (Table 7) and therefore result in a positive population trajectory (Figure 13). If the oiling mortality on the offshore banks is assumed to be lower and only 5 %, then a 10 times lower mortality (0.5 %) would result in a 0.9 % total mortality due to oiling, and a lambda of 1.008. However, the proportional increase to the unchanged shipping is lower for the second rerouting scenario.

Table 7. Estimated growth rate (Lambda) and predicted female population size in 2026, for two scenarios assuming 10 times lower Long-tailed Duck oiling mortality on the Swedish offshore banks. The last column shows the predicted % change in population size in between the rerouting scenario in comparison with the baseline (no change in shipping).

Model	Lambda	Estimated pop. size in 2026	% population increase in 2026 compared to unchanged shipping
M2 + rerouted shipping	1.006	470,523	+30
M3 + rerouted shipping	1.008	514,784	+12

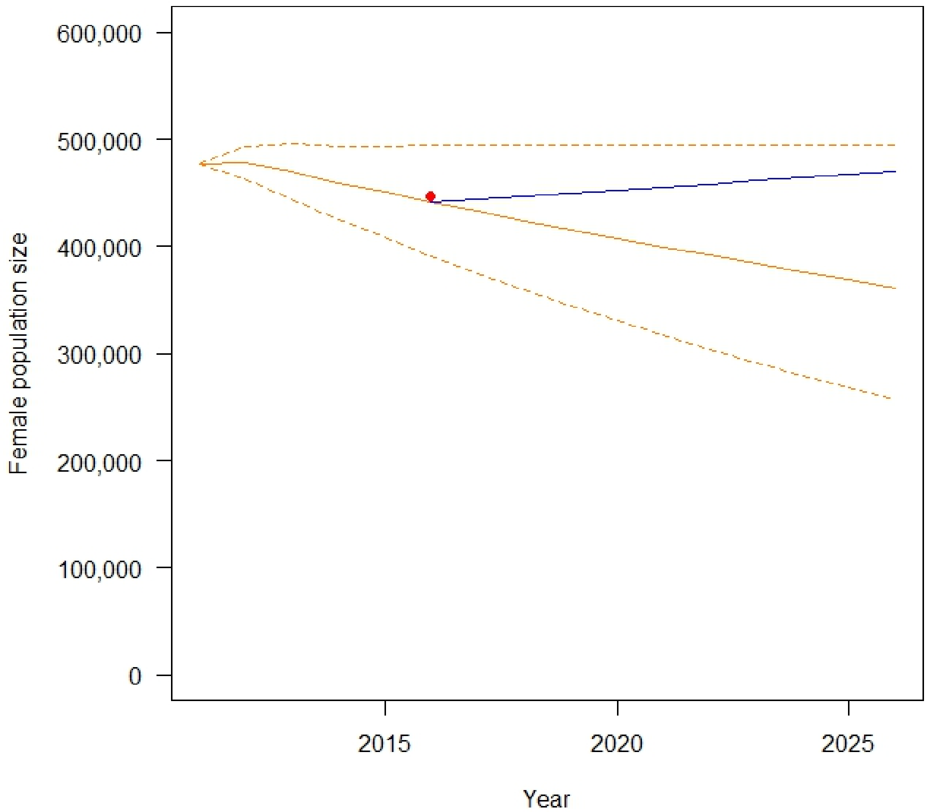

Figure 13. Population trajectories predicted by the LPM model. The orange line indicates the population trajectory predicted according to submodel M2 with a lambda of 0.98. The dashed lines indicate the model fitted with either upper or lower standard error around the mean productivity rate estimated in winter and spring in Larsson (submitted). The red dot indicates observed numbers of Long-tailed Duck during the survey conducted in 2016. The 2016 number is extrapolated to the total female population based on assuming the Swedish offshore bank population being 25 % of the total population.

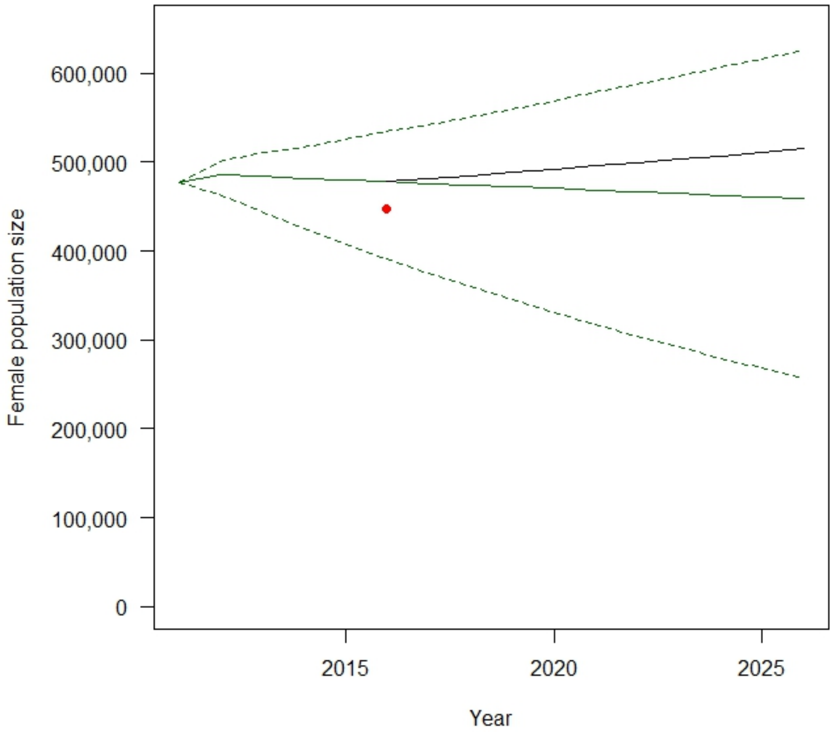

Figure 14. Population trajectories predicted by the LPM model. The green line indicates the population trajectory predicted according to submodel M3 with a lambda of 0.996. The dashed lines indicate the model fitted with either upper or lower standard error around the mean productivity rate estimated in winter and spring in Larsson (submitted). The red dot indicates observed numbers of Long-tailed Duck during the survey conducted in 2016. The 2016 number is extrapolated to the total female population based on assuming the Swedish offshore bank population being 25 % of the total population.

6.3 Question 3 - Effect of shipping on harbour porpoises

6.3.1 Effect of the current anthropogenic ambient noise on porpoises

The range of sound level of ambient noise at 2000 Hz at the study site was between 98 and 130 dB. Overlap of the areas with ≥110 dB is shown in Table 8 and Figure 15. The overlap of the most suitable habitat class (class 10) with the noise level triggering avoidance behaviour (110 dB) was lower during the summer (30.6 %) than winter (36.6 %) but for the remaining habitat classes the tendency was opposite. None of the areas were exposed to sound level ≥ 170 dB indicating risk of TTS effects on harbour porpoises.

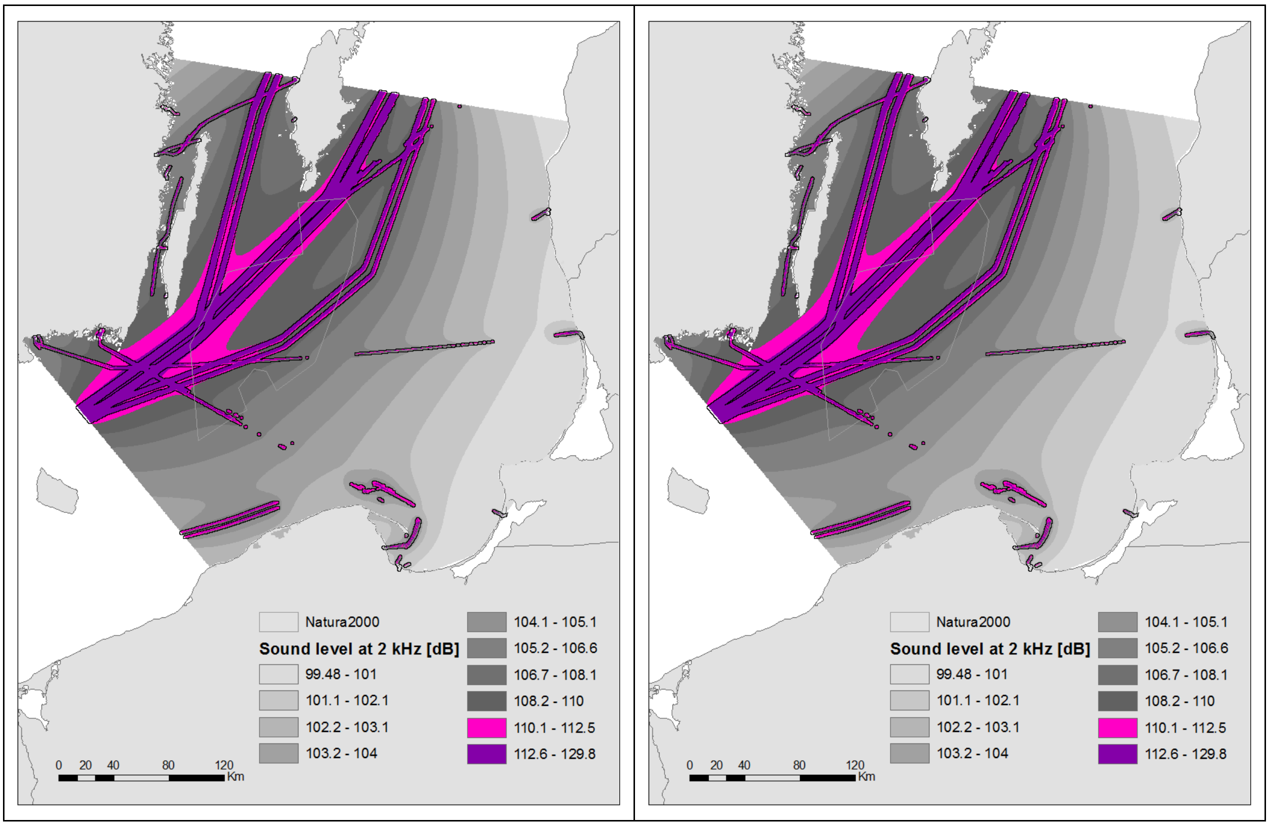

Figure 15. Quantiles of sound level at 2000 Hz calculated for summer (left panel) and winter (right panel). High shipping routes are depicted by black polygons, including 1km buffer around them. Pink areas represent areas of sound level > 110 dB: level triggering avoidance behaviour.

Table 8. Percentage overlap between habitat classes and areas with sound level over thresholds triggering avoidance behaviour (110 dB) and temporary threshold shift (170 dB).

	Summer		**Winter**	
Habitat class	dB		dB	
	110	**170**	**110**	**170**
10	30.6	0	36.6	0
20	43.6	0	36.1	0
40	48.2	0	24.1	0
60	13.3	0	12.4	0
80	8.1	0	5.8	0

6.3.2 Effect of the current ship presence on harbour porpoises

The 10 % quantiles of AIS (‘High-traffic’ areas) ranged from 98 to 7300 ships during summer and 110 to 6100 during winter (Figure 16). The highest traffic is at the south-eastern part of the study site along the traffic route just before the divergence of West Gotland and “Original route” going through the study area north of the offshore banks.

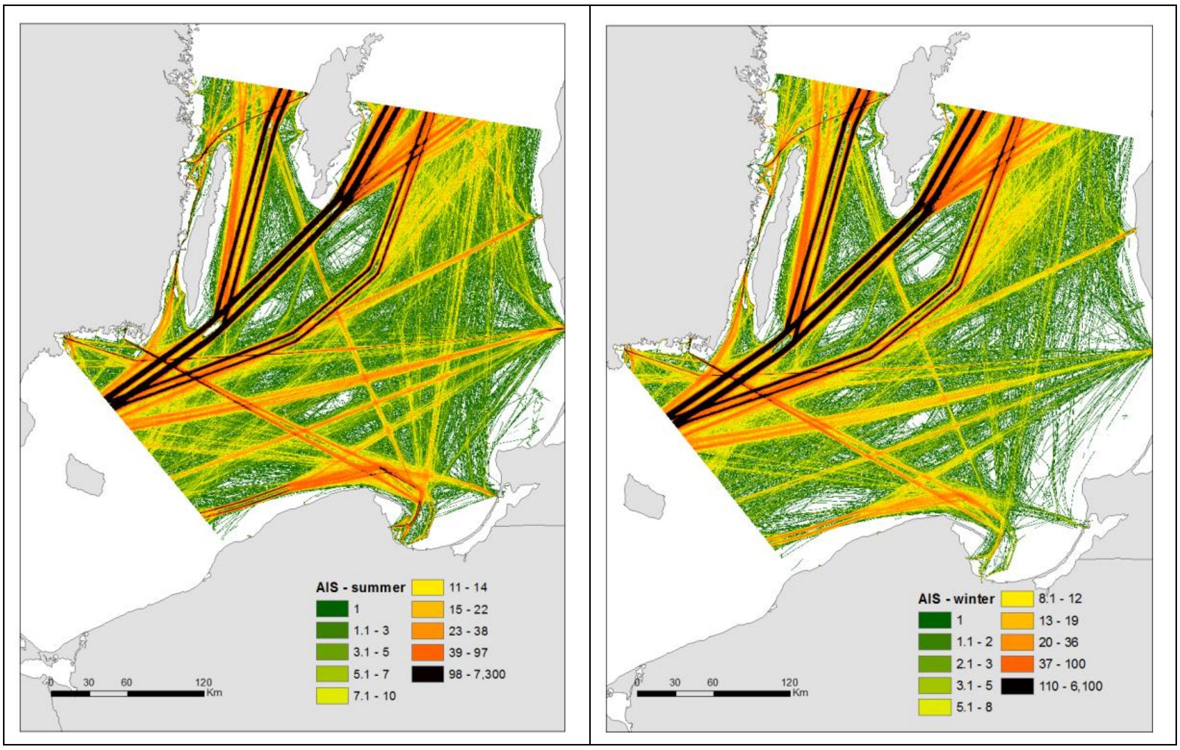

Figure 16. 1 0% - interval quantiles of AIS data for summer (left panel) and winter (right panel) 2015. AIS represent a sum over tankers, cargo and passenger boats.

The areas of high-traffic and the overlap between them and habitat classes is shown in Figure 17 and Table 9. The overlap varied between 5 and 31 % and was on average higher for summer (mean 19 %) than winter (16 %). 15 % (summer) and 23 % (winter) overlap was found between the most suitable habitat class and areas of high-traffic.

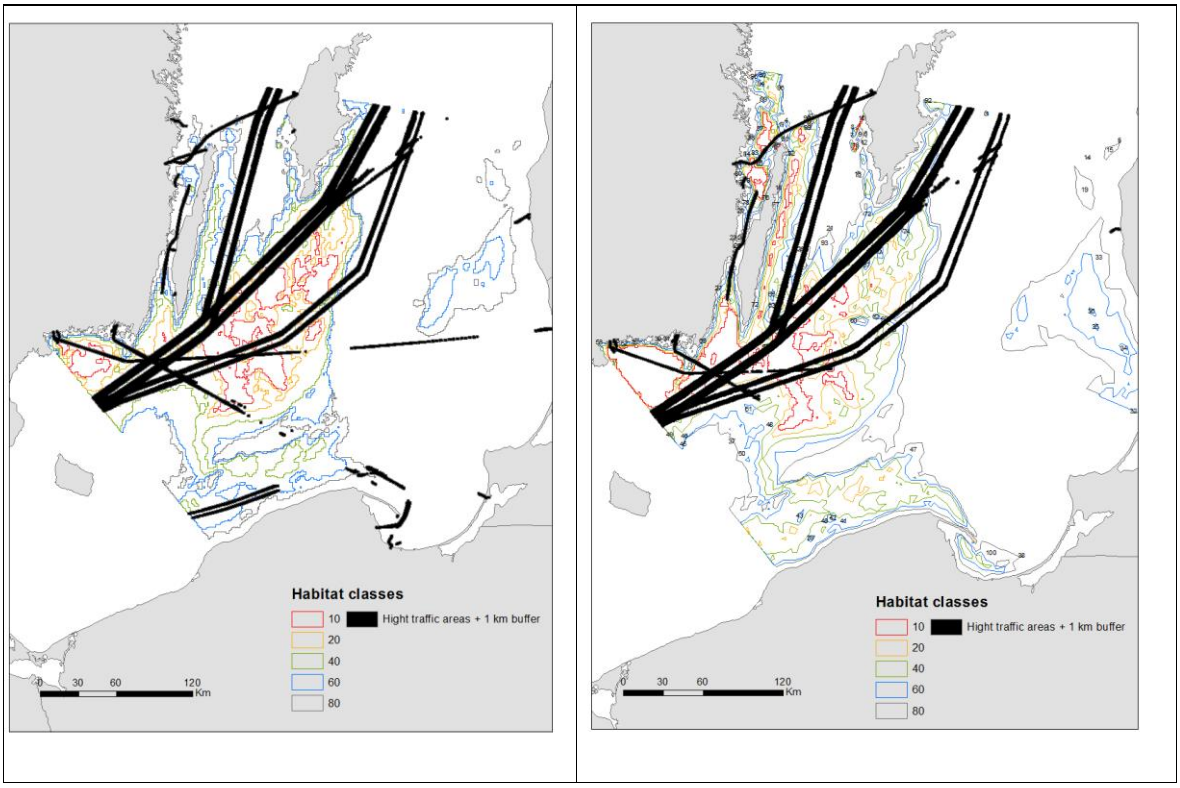

Figure 17. High-traffic areas and their overlap with habitat classes for summer (left panel) and winter (right panel).

Table 9. Percentage overlap between habitat classes and areas with high-traffic including 1 km buffer.

Habitat class (quantiles)	Overlapping area (%)	
	Summer	**Winter**
10	15.1	23.2
20	27.2	21.6
40	31.1	18.1
60	9.03	10.0
80	13.0	5.5

6.3.3 Population level consequences of anthropogenic disturbance

Little is known about the effect of anthropogenic disturbance on the harbour porpoise population dynamics. Disturbances can affect animal foraging and thereby have an effect on individual fitness and population survival by the exclusion of animals from high-quality foraging areas and by the net energy losses associated with fleeing from disturbances (e.g. Baveco et al. 2011, Kerley et al. 2002). Even if avoidance and masking may affect foraging time, prey detection and energy balance only temporarily, their cumulative and combined effect may be significant. Nabe-Nielsen et al. (2014) estimated cumulative effect of disturbance related to bycatch and noise from ship traffic and wind farms on the Belt Sea population of harbour porpoises. They found that although noise from the wind farms and ships does not result in population declines, the porpoise population is sensitive to the speed at which food recovers after being depleted. If food recovers slowly, the effect of ships are estimated to have a significant negative impact on the population. Disturbance impacts like the ones assessed for parts of the most suitable areas to porpoises south of Gotland, may seem negligible on the short term. However, depending on the rate of prey replenishment they could potentially translate to serious long-term effects within the overlapping zones (roughly 20-30 % of most suitable areas), with impacts on both individual fitness and population dynamics (Bejder et al. 2006, Hermannsen et al. 2014, National Research Council 2003).

6.4 Question 4 – effect on harbour porpoises after redistribution of shipping

6.4.1 Effect of anthropogenic ambient noise of reallocated traffic on harbour porpoises

The range of sound level of ambient noise at 2000 Hz after route relocation at the study site was between 97 and 129 dB. Overlap of the areas ≥110 dB is shown in Table 10 and Figure 18. Relocation of route resulted in decrease of average overlap from 28 % to 12 % for summer and 23 % to 12 % (9 % for scenario with no change in width) in winter. Scenario with change in width of the new route resulted in increase in area of overlap and this increase was more pronounced for winter. None of the areas were exposed to sound level ≥170 dB indicating risk of TTS effects on harbour porpoises.

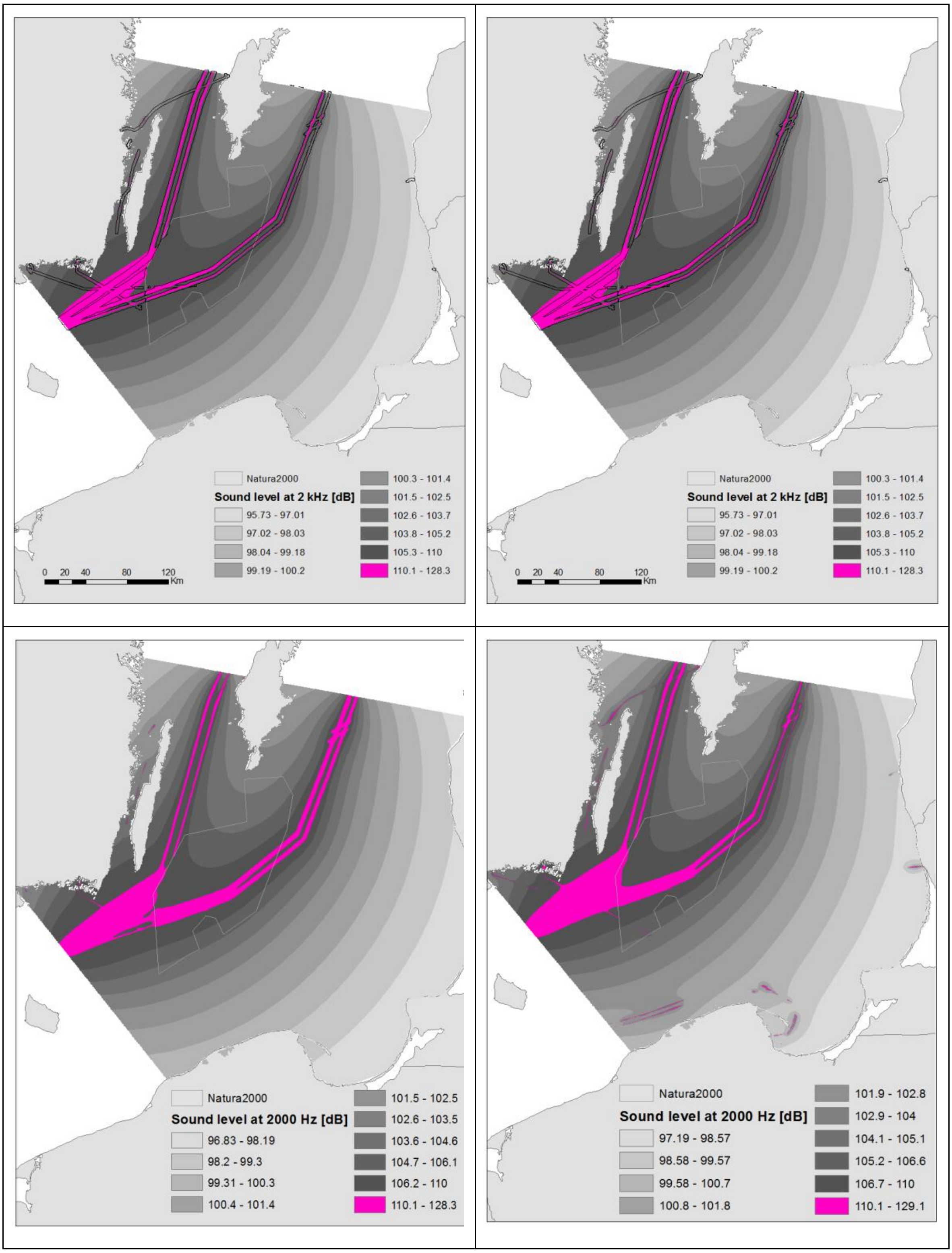

Figure 18 Quantiles of sound level at 2000 Hz calculated for summer (left panels) and winter (right panel) after relocation of high-traffic from Original to Deep Water route. Results are shown for two scenarios: no change in width of the new route (upper panels) and with width change (lower panels). New, relocated shipping routes are depicted by black polygons including 1 km buffer around them. Pink areas represent areas of sound level >110 dB: level triggering avoidance behaviour.

Table 10. Percentage overlap between habitat classes and areas sound level over thresholds triggering avoidance behaviour (110 dB) and temporary threshold shift (170 dB). Results are shown for two scenarios: with and without width change of the new route.

Habitat class	Overlapping area (%) – no width changed				Overlapping area (%) - width change			
	Summer-110	**Winter-110**	**Summer-170**	**Winter-170**	**Summer-110**	**Winter-110**	**Summer-170**	**Winter-170**
10	12.0	20.0	0.0	0.0	12.1	23.2	0.0	0.0
20	19.1	15.7	0.0	0.0	19.2	19.8	0.0	0.0
40	24.6	8.4	0.0	0.0	24.7	10.4	0.0	0.0
60	3.1	2.4	0.0	0.0	3.2	4.5	0.0	0.0
80	1.7	1.5	0.0	0.0	1.7	1.9	0.0	0.0

6.4.2 Effect of reallocated ship presence on harbour porpoises

The traffic intensity along the “Deep Water traffic route” after reallocation from the Original to Deep Water route for two scenarios is depicted in Figure 19.

The area of best habitat class overlapping with high-traffic areas decreased for summer and winter months after route reallocation for both scenarios (Table 9, Table 11). After relocation, the average overlap for all habitat classes decreased both for summer (from 19 % to 12 %) and winter (from 16 % to 10 %). The differences in area overlap between two scenarios (with/without width change) were negligible (Table 11).

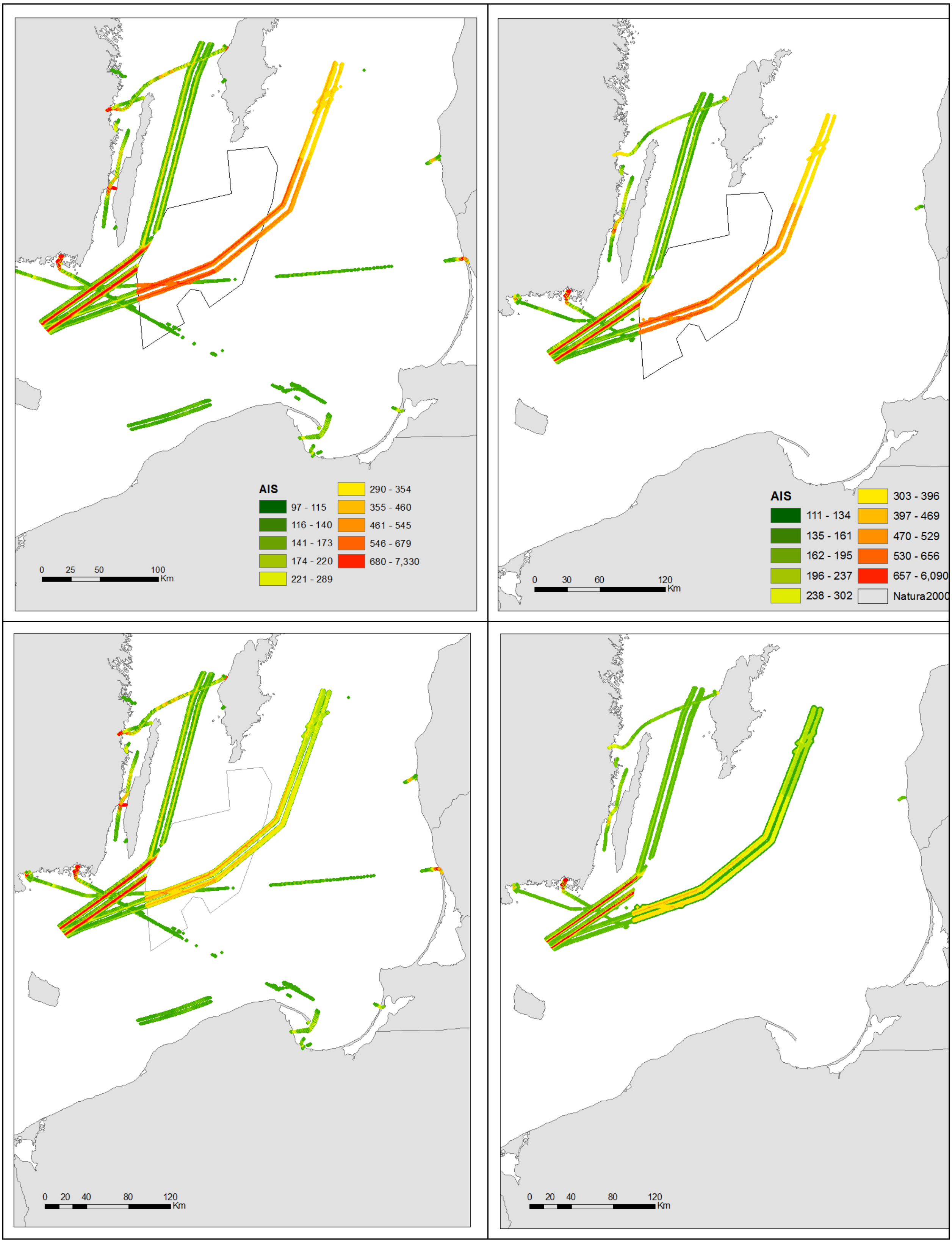

Figure 19. Traffic intensity along the new route for summer (left panels) and winter (right panels) for two scenarios: no change in width (upper panels) and increase in width (lower panels) of the new route. Black polygon depicts Natura 2000 area.

Table 11. Percentage overlap between habitat classes and areas with high-traffic including 1 km buffer after reallocating traffic from Original to Deep Water shipping line. Two scenarios were analysed: with and without width change of the new route.

Habitat class	Overlapping area (%) - width changed		Overlapping area (%) - no width change	
	Summer	**Winter**	**Summer**	**Winter**
10	11.9	20.5	11.8	18.1
20	16.9	15.3	17.0	12.9
40	20.7	11.6	20.3	10.3
60	4.7	7.1	4.6	5.4
80	4.2	4.5	4.2	4.0
Mean	11.7	11.8	11.6	10.1

6.5 Question 5 – Effect of redistribution of shipping on harbour porpoises in the enlarged Natura 2000 area

The area of overlap between high-traffic and the enlarged Natura 2000 area decreased after route reallocation by 10 % for summer and 5 % for winter and this decrease was comparable between scenarios (Table 12). Decrease of area overlap between Natura 2000 and area impacted by noise over 110 dB was even more pronounced (28 and 26 % for summer and winter respectively) but comparable between scenarios.

Table 12. Percentage overlap between Natura 2000 area and areas with high-traffic including 1 km buffer (for two scenarios: with and without width change if the new route) and between Natura 2000 and areas with noise level exceeding 110 and 170 dB thresholds at current routes and after relocating traffic from Original to Deep Water shipping line.

			Current route	**Relocated route**
Overlap	**No width change**	**Summer**	25.0	15.4
		Winter	21.4	15.5
	Width change	**Summer**	25.0	15.7
		Winter	21.4	16.0
Noise	**No width change - 110 dB**	**Summer**	41.4	13.7
		Winter	31.5	6.0
	Width change - 110 dB	**Summer**	41.4	13.9
		Winter	31.5	13.9
	No width change - 170 dB	**Summer**	0.0	0.0
		Winter	0.0	0.0
	Width change - 170 dB	**Summer**	0.0	0.0
		Winter	0.0	0.0

7 Discussion and Conclusions

7.1 Long-tailed duck

By using a population model (LPM model) together with the available information on Long-tailed Duck life history parameters, we have been able to assess the relative importance of the different parameters. The LPM model clearly showed that adult survival is the most sensitive parameter, i.e. a change in survival has the proportionally highest influence on the lambda (growth rate) and thereby the overall population trajectory. This has also been shown by others (e.g. Schamber et al. 2009 and Larsson submitted). There is little quantitative information available on the Baltic Long-tailed Duck population parameters. Particularly on lower level fecundity parameters, however the immature male/adult female ration provided by Larsson (submitted) represents highly useful information. This ratio is also in correspondence with values estimated in North America, based on lower level fecundity parameters as clutch size, nesting success etc. Further, there is only two total population estimates available based on large-scale surveys, although the trend is confirmed by counts of migrating birds and other small-scale surveys (see Hearn et al 2015 and reference within). Therefore, although there is a large uncertainty coupled with these numbers, they can be considered to provide a description of the change in population dynamics. Based on the estimated fecundity (or productivity) and the population trend observed it was possible to estimate a survival rate. In addition, because some information is available on mortality caused by recurrent operational oil discharges and also a potential trend in oiling mortality, it was possible to quantitatively estimate a potential mortality effect related to shipping in the area of interest on the Swedish offshore banks south of Gotland. It was estimated that even if there is more than a 50 % reduction in oiling mortality (from 11 % to 5 %) compared to the beginning of 2000, the mortality only on the offshore bank is >1 % of the total population annually. This value is not including mortality or a reduction in productivity related to habitat displacement. It is assumed that the effect of habitat displacement is minor at the moment when the population size is rather small.

However, habitat displacement might still contribute to the additive mortality. The effect of habitat displacement might be a larger issue in the future, if large-scale offshore infrastructures in Long-tailed Duck habitats are realised.

What would then happen if ships would be rerouted from the shipping lane passing north of the banks to the southern deeper shipping lane? Based on the estimate from by Forsman (2017), where the authors suggested a 10 times lower exposure to oiling if ships are rerouted, it was estimated that a rerouting would result in an increase in lambda, in the best case scenario from 0.996 to 1.008. This means from a negative to a positive growth rate. When these two scenarios (unchanged vs changed shipping routes) are projected into the future, the LPM model predicted a 12 % increase in population size. The model results therefore suggest that a rerouting of shipping from the shipping lane crossing the study area north of the banks to the Deep Water lane south of the banks could have a significant effect on the population dynamics. According to the LPM model, it could potentially result in that the declining population would start increasing instead.

7.2 Harbour porpoises

Based on the assessment, around 1/3 of high quality harbour porpoise habitats are influenced by noise level triggering avoidance behaviour of porpoises during summer and even more (36 %) during winter. Even if values for winter are higher, the summer distribution of porpoises is more constrained and the highly suitable area is mostly within the newly designated Natura 2000 area. Therefore, an effect of shipping during summer may be more pronounced despite a smaller area being affected. If only overlap between shipping and highly suitable areas is considered, there is a higher overlap during winter in comparison with summer. No TTS effects on porpoises were estimated for any of the scenarios.

According to our analyses, 31-41 % of the Natura 2000 area is currently affected by sound triggering avoidance behaviour. Only a sound level at 2000 Hz was analysed, yet larger vessels can produce sounds with higher frequencies, which will have even more pronounced effects on porpoises.

The assessment also suggests that a rerouting of ships from the shipping lane crossing the area north of the banks to the Deep Water lane would result in a decreased area of impact. This change is more pronounced during winter than summer, if only overlap with shipping is taken into account. A decrease in the affected area regarding noise is more pronounced during summer than winter. As summer represents the calving season, a reduction in impact might be particularly beneficial during this season. However, it is important to note that the analyses of the dispersal of underwater sound are simple and “rough” and should only be assumed to be approximate. When converting grids and calculating sound impacts, there was an artificial reduction (due to the analyses method) in shipping lane grid cells, which mean that the sound levels around the Deep Water lane is estimated slightly too low. However, the results give an indication of the direction of change, i.e. the estimated reduced impact can be assumed realistic.

An interesting and important result of the analyses are that the width of the shipping lanes matter. So even if the ships are not rerouted but the shipping lanes made as narrow as possible the impact on porpoises would be reduced, and if the ships would be rerouted keeping the width of Deep Water lanes the same as before reduces the impact more that making the lane wider.

In terms of consequences at the population level the estimated behavioural and masking effects may lead to cumulative and combined effects which enhances the risk of reduced individual fitness and population survival by habitat displacement (Kerley et al. 2002, Baveco et al., 2011). Whether such effects translate into serious local impacts depend on the status of local prey populations, and in particular the frequency of prey replenishment (National Research Council 2003, Bejder et al. 2006, Hermannsen et al. 2014).

8 Knowledge gaps and recommendations for future studies

8.1 Long-tailed Duck

There are still many knowledge gaps regarding the Long-tailed Duck population. An increase in data and knowledge regarding all aspects of the population can be considered to be needed. However, a few things considered to be particularly useful for advancing the understanding of important drivers behind the population dynamics and improve the LPM model are listed below.

A potential way to improve the model is to consider the population parameters as stochastic instead of deterministic. This could be relevant, as the parameters are highly variable in time, stochastic, and the model results from a stochastic model could therefore be considered to be more realistic. The main difference between a deterministic and a stochastic model is that a deterministic model describes a potential effect of a management action whereas a stochastic model provides a probability of realizing that effect (Flint 2015).

Another aspect for which very little information exists are movements and connectivity of the Long-tailed Duck populations on the Swedish offshore banks. Do the birds utilise all of the banks, are they connected to the high-density areas in the Gulf of Riga and the Pomeranian Bay located within 300 km distance or do they have small home ranges and display a high site fidelity to a specific bank? These are key questions to answer if we want to be able to understand how different anthropogenic pressures might affect the population dynamics in the future. If the birds on the offshore banks also utilize coastal areas and are therefore also impacted by pressures in the coastal areas, like reductions in food supply caused by a successful implementation of the Baltic Sea Action Plan and improved eutrophication control, the population trajectories following a change in shipping patterns south of Gotland may develop markedly different than predicted by this study. Limited information is also available about the energy budgets in the study area. In Fehmarnbelt, in southern Denmark, Long-tailed Ducks spend up to 60 % of their time under water, i.e. they are forced to feed a large proportion of the time (FEBI 2013). This could mean that a displacement from important foraging areas can have a relative large impact on the population. However, it might be that Long-tailed Ducks on the offshore banks are not forced to feed with the same intensity. These questions could be studied and answered with the help of telemetry tools, by tagging birds on the banks and following their movement throughout the wintering season. It could potentially even be possible to attach dive loggers on the birds, which would reveal diving behaviour and thereby inform us about food availability (habitat quality) in different areas. Avoidance of shipping lanes and potential habitat displacement would also be revealed by a telemetry study. Telemetry data would provide highly valuable information, which would help in estimating impacts of anthropogenic pressures and management actions on the population dynamics of Long-tailed Ducks.

Conducting more surveys would provide us with enhanced information on population size and variability during a wintering season in the core area on the Swedish offshore bank. This would help us reducing uncertainty regarding the number of Long-tailed Ducks currently utilising the Swedish offshore banks.

8.2 Harbour Porpoise

Future studies or improvement of the current study regarding harbour porpoises could be undertaken by improving the sound modelling by utilising a more advanced sound modelling approach for example the dedicated DHI numerical noise modelling software, the Underwater Acoustic Simulator (UAS). Instead of using cumulative AIS data (i.e. not actual number of ships passing during a given time snapshot), original AIS data could be used to analyse the impact of sound during the calving period. The sound dispersion model could be made more detailed, which would improve both the estimate of traffic intensity and the assessment of the importance of the width of the shipping lane.

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