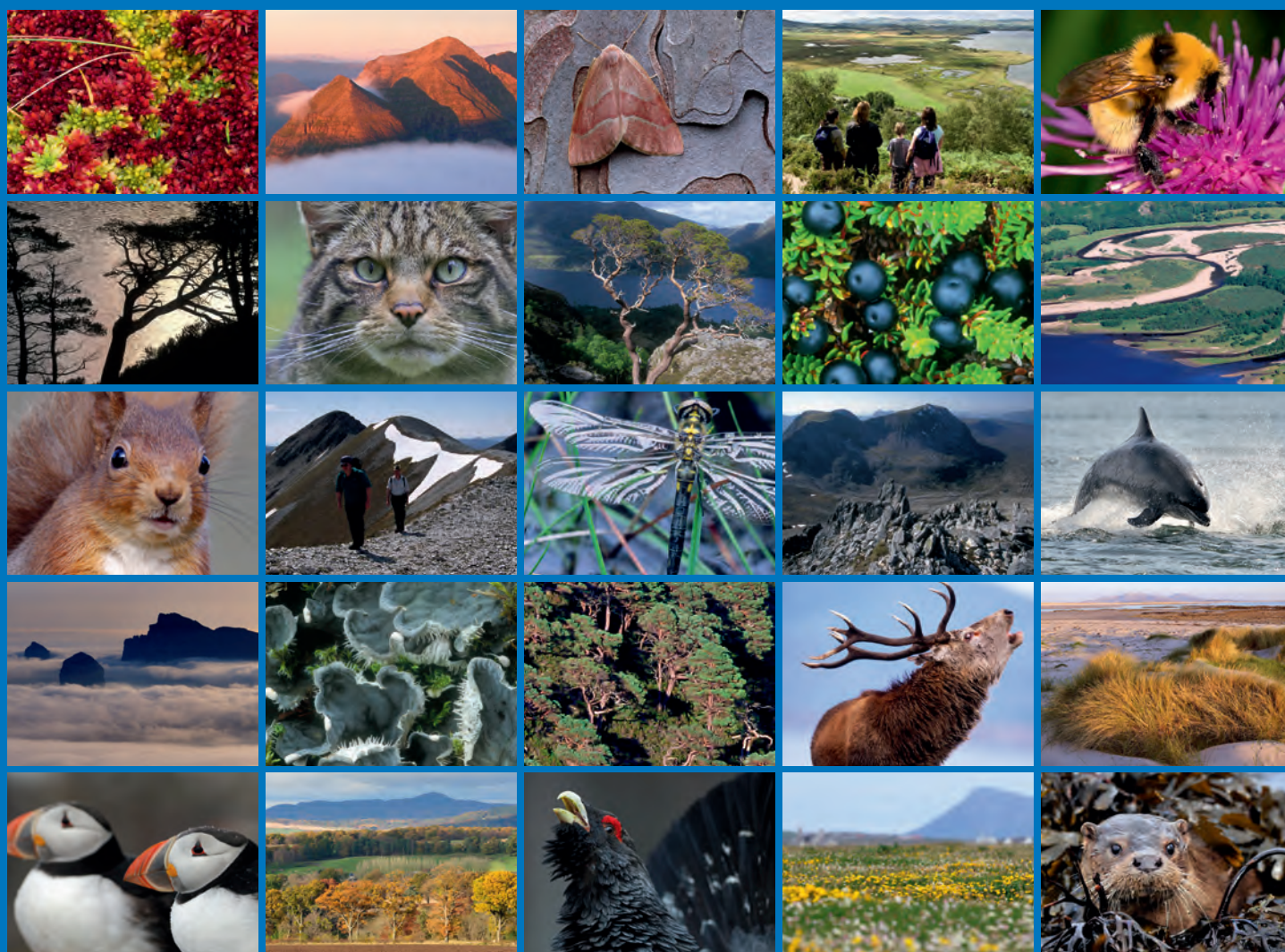


A diving bird collision risk assessment framework for tidal turbines





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COMMISSIONED REPORT

Commissioned Report No. 773

A diving bird collision risk assessment framework for tidal turbines

For further information on this report please contact:

Chris Eastham
Scottish Natural Heritage
Great Glen House
INVERNESS
IV3 8NW
Telephone: 01463 725000
E-mail: chris.eastham@snh.gov.uk

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COMMISSIONED REPORT

Summary

A diving bird collision risk assessment framework for tidal turbines

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Marine renewables; collision risk; diving birds; tidal turbines.

Covering statement

This report describes an approach for assessing the collision risk of diving birds with marine renewable energy devices, known as the exposure time population model (ETPM). The approach explores the collision rate required to achieve a critical level of additional mortality by estimating (i) thresholds of additional mortality for the population at risk of collision and (ii) the potential time that each individual within the population is at risk of collision. A judgement is then made as to whether the maximum acceptable collision rate is likely to occur or not.

The ETPM approach provides a broad based assessment of the potential impact of collisions with tidal turbines on populations of diving birds. Given the limited knowledge base and poor understanding of the underwater movements of diving birds and their behavioural responses to underwater devices, this approach is considered an appropriate and useful method for assessing collision risk of diving birds.

Apart from the ETPM, there are a number of other models used to assess collision risk of marine wildlife, namely the adapted Band / SNH collision risk model and the SRSL encounter rate model. We currently do not favour any one model when undertaking a collision risk assessment. All of the available models are likely to have imperfections, and the accuracy of the model predictions is dependent on the quality of the input data. In particular scenarios, the preferred model choice may differ, for which an appreciation of the model parameters and mathematical functions is required. Ultimately, however, the results of any assessment of overall risk are likely to be determined by the assumptions made about the animals' ability to avoid collisions. At this early stage of our understanding it may be necessary to take a relatively precautionary approach in collision risk assessment, including model design and choice, with a view to improving these options as real-life observations improve our understanding of how animals behave in the water and around these developments.

We advise that developers establish early contact with Marine Scotland and SNH to discuss options for undertaking collision risk assessment in connection with any planned marine

renewable energy development. Further guidance on underwater collision risk assessments for marine wildlife is currently in development by SNH.

Background

Marine tidal energy schemes are likely to make a substantial contribution to the mix of future energy sources within Scotland and the UK, but their environmental impacts are poorly understood. For diving seabirds, collisions with tidal turbines represent a potentially important way in which tidal energy developments may cause population-level impacts.

Estimating collision rates for diving birds is problematic due to poor knowledge of underwater swimming behaviour and of the ability to evade turbines in the underwater environment. This is particularly the case, given that the estimation of collision rates may be highly sensitive to the assumed avoidance rate, for which there is considerable uncertainty for diving birds interacting with tidal turbines. This report describes an approach to assessing the collision risk posed by tidal turbines that is based upon estimating exposure time and which does not rely on direct estimation of collision rates.

Main findings

- This report sets out the rationale for the ETPM approach, detailing the underlying theory from which a ‘required’ collision rate can be derived from estimation of (i) thresholds of additional mortality for the population at risk of collision (via population modelling) and (ii) the potential time that each individual within the population is at risk of collision (via exposure time modelling).
- The parameters from which exposure time is estimated are described, along with the means by which these parameters can be derived from a combination of data extracted from the available literature and obtained from site-based measurements.
- The modelling approach is illustrated using two example species (European shag *Phalacrocorax aristotelis* and common guillemot *Uria aalge*) at an unspecified proposed tidal development in the Pentland Firth, northeast Scotland. These examples indicate that the collision rate required to exceed the ‘acceptable’ threshold of additional mortality is considerably lower (and hence the collision risk higher) for shag than for guillemot at this site, in keeping with the higher on-site densities and smaller source population of the former species.
- Exposure time estimates are shown to be relatively sensitive to variation in several of the input measurements, some of which have considerable uncertainty associated with them. This has consequences for the derivation of the ‘required’ collision rate, although in some situations at least there is likely to be scope to reduce uncertainty in some measurements through targeted on-site data collection.
- The ETPM approach provides a broad based assessment of the potential impact of collisions with tidal turbines on populations of diving birds. This level of assessment is considered to be appropriate given the limited knowledge base and poor understanding of the underwater movements of diving birds and their behavioural responses to underwater devices, including their ability to evade the rotating turbines.

For further information on this project contact:

Chris Eastham, Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.

Tel: 01463 725000 or chris.eastham@snh.gov.uk

For further information on the SNH Research & Technical Support Programme contact:

Knowledge & Information Unit, Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.

Tel: 01463 725000 or research@snh.gov.uk

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1. INTRODUCTION

1.1 Tidal energy schemes and birds

Both the Scottish and UK Governments have made commitments to source increasing amounts of their total energy needs from renewable sources, such as wind, wave and tidal (e.g. The Scottish Government 2011). Marine tidal currents are considered to offer greater predictability for energy generation than other forms of renewable energy such as wind farms, due to the fact that the tides are not affected by weather conditions. As such, tidal energy schemes are likely to make a substantial contribution to the mix of future energy sources.

Methods for extracting energy from tidal sources are currently being trialled with a view to full scale developments being installed in the near future. However, there is a lack of knowledge on how such developments will impact marine wildlife, and on the ways in which marine wildlife will interact with the turbine devices used to harness tidal energy. This includes effects on seabird species, for which Scotland and the UK hold populations of considerable international importance (Mitchell *et al.* 2004, McCluskie *et al.* 2013). Collision with tidal turbines by diving birds (causing death or injury) is one of the potential ways in which such developments may affect seabird populations. Consequently there is a requirement to develop analytical methods suitable for assessing the potential impact of underwater devices, such as tidal turbines, on seabird species.

1.2 Collision risk considerations

In relation to collision risks, impacts on bird populations have been identified previously as an issue of concern for other types of renewable energy developments, particularly as a consequence of the expansion in the number of wind farms, both on- and off-shore (Drewitt & Langston 2006). Thus, substantial work has been undertaken on this topic, albeit in a different environment, with birds considered to be potentially at risk from collisions with moving turbine blades at wind farms. As a consequence of this, various collision risk modelling frameworks have been developed to attempt to estimate the mortality levels which could occur through collisions with wind turbines (e.g. Tucker 1996), including those used widely for this purpose within the UK (Band *et al.* 2007, 2012). Inputs to such collision risk models (CRMs) include observations of bird flight paths through the proposed wind farm site, along with species specific bird metrics such as average flight speed, body length and wingspan, and wind farm specific variables such as rotor speed and diameter. The outputs from the model are predictions of the number of collisions per unit time for each species, with an avoidance rate applied to these predictions to account for the fact that most individuals will avoid flying through the turbines. The issue of estimating the avoidance rate is problematic and reliable data on avoidance rates are lacking for the vast majority of species (SNH 2010). Avoidance rates are likely to be species specific (depending upon a range of factors such as behavioural response and manoeuvrability in flight), although early assessments of collision risk using the 'Band model' adopted a fixed 95% avoidance rate for all species. With time (and emerging evidence of collision rates from operational wind farms), the recommendation has been to increase this value for most species (SNH 2010).

Collision risk estimates derived from the 'Band model' are highly sensitive to variation in the avoidance rate, and the effect of variation in this parameter on the final collision risk estimate obtained may be two orders of magnitude greater than that for equivalent variation in any other model parameter (Chamberlain *et al.* 2006). Therefore, accurate estimation of avoidance rates is critical for robust estimation of collision risk, but the resource-intensive, and potentially site-specific, nature of the work required to estimate avoidance rates (e.g. Whitfield & Madders 2006) means that there remain relatively few species for which this has been achieved. Furthermore, problems with CRMs may extend beyond the accurate estimation of avoidance rates. Ferrer *et al.* (2011) found no clear relationship between the

risk predicted by EIA studies and the actual recorded bird mortality across 20 terrestrial wind farms. This was because the linear relationship assumed between the estimated number of birds at rotor height and the actual recorded bird mortality, did not hold.

In relation to assessing the collision risk to seabirds from tidal turbines, the focus is primarily on deep-diving species such as auks, although in some situations devices may be located at or near the surface, so that impacts could occur on a much wider range of species. The nature of movements of diving birds is less well understood than for birds in flight, with little known about the behavioural responses to underwater devices and the ability to evade rotating turbines as birds pass through the device (McCluskie *et al.* 2013). Also, several of the assumptions made within the 'Band model' (e.g. that birds travel through the rotor swept area on a horizontal trajectory) are less likely to be met by birds swimming underwater than by birds in flight. An additional complication is that, unlike wind turbines which are essentially very similar in design, the range of devices currently being proposed for tidal (as well as other marine) energy generation is extremely varied. Therefore, there are considerable challenges in assessing the collision risk from tidal turbines (and other underwater devices) to diving seabirds, which are likely to make collision risk difficult to estimate reliably using modelling frameworks of the types developed for wind turbines.

In this report a method for assessing collision risk from tidal devices is presented, along with the rationale underlying the adopted approach, and consideration of the parameters that would be required to adapt the 'Band model' to this purpose. The proposed method is illustrated with example data collected at a site with high potential for the development of tidal stream devices, and an assessment is made of the sensitivity of the outputs to variation in the different input parameters. Although the specification under which this work has been carried out identified tidal turbines as the generic device on which to base the assessment, as will be seen, the approach proposed is not device-specific and would therefore be equally applicable to assessing impacts from other forms of underwater devices.

2. REQUIREMENTS FOR ADAPTING THE BAND WIND FARM COLLISION RISK MODEL TO TIDAL TURBINES

In this section, consideration is given to aspects of the 'Band model' (Band *et al.* 2007) that are of key importance in the estimation of collision risk, and to the modifications required to adapt the approach to tidal turbines and data required to populate it for that application. This is considered specifically in relation to the on-shore model but applies equally to the offshore model (Band 2012).

Ideally a CRM provides a modelling environment within which the probability of a bird colliding (fatally) with a device could be predicted based on species-specific characteristics (recorded at the proposed site) and device specific turbine characteristics. This approach would be broadly comparable with the wind turbine CRM.

The bird data required for such a model include:

- Dive depth, duration and frequency.
- Distance travelled during dives and the underwater space occupied by the dive.
- Swim speed (ascent, descent, horizontal and burst/escape).
- Underwater manoeuvrability, awareness and evasive response (avoidance).
- Activity levels in relation to the annual cycle, and tidal state and time of day.

While turbine data would be device specific, there would probably be sufficient generic data (e.g. turbine diameter, number of blades and rotation speed) available to estimate typical

values. Additional characteristics which might also be important would be related to the location, and would include factors such as tidal flow speeds, water depth and device depth.

Outputs from a tidal CRM would be estimates of the number of fatal collisions per unit time per turbine. These estimates could then be reviewed in relation to the effective population size of the target species to determine the potential impact on the population level at a range of spatial scales including potential effects on SPAs.

The challenge for this approach would be to collect robust estimates for the bird parameters listed above. Of these, the first five (dive depth, duration, frequency, distance travelled underwater and swim speed) are equivalent to the flight data collected for wind farm assessments. Underwater manoeuvrability, awareness and evasive responses can all be considered as aspects relating to avoidance behaviour.

It is also worth noting that the wind farm collision model is based on the premise that birds pass through the site on transit flights (although there is allowance for indirect flights, i.e. not simply straight lines). However, not all diving birds are undertaking the equivalent of transit flights underwater, but rather are actively foraging in the volume within which the devices will be sited (with diving birds pursuing fish prey perhaps being more analogous to hirundine flights in pursuit of flying insects, albeit that the latter travel at much higher speeds). For species which forage at the sea bed like common eider *Somateria mollissima*, the 'transit' analogy may be more reasonable, since the period during which they will be at risk of collision will be limited to the journeys to and from the sea bed.

Generic information for dive metrics has been obtained for some species (Table 1), but there are indications of high between-study variability in some of these data and the extent to which these data are site-specific remains unclear. The distance travelled underwater may be a critical parameter and this is probably less well known, although may be closely related to the rates of depth, duration and dive frequency (Wanless *et al.* 1997a).

Measures which will influence the effective avoidance rate are likely to be even less well understood. One possible indirect approach to deriving these is on the basis of characteristics of the prey types being selected by each species (e.g. pelagic fish, benthic fish, benthic invertebrates), although this might only provide an index of the ability to evade collision when close to turbine rotors, and would be unlikely to reflect any trait to avoid close proximity to the devices themselves. Furthermore, using an adaptation of the 'Band model' is likely to mean that the results of CRM for tidal turbines will suffer from the same sensitivity to variation in the estimated avoidance rate as do results for wind farms. Given the considerable difficulty in estimating underwater avoidance rates, it is proposed that an alternative approach to determining possible impacts should be considered.

Table 1. Summary of reported dive metrics for species considered at potential risk from underwater turbines.

Species	Mean dive depth	Maximum dive depth	Dive duration	Dive entry and shape	Prey type	Source
Common guillemot	10m (50% dives <6m and 90% dives <22m) (Norway)	37 (Norway) 50m (Norway) 53m (Scotland)	Mean 39s; max. 119s (Norway)	Surface diver. U-shaped (mean bottom dive duration of 19s) (Norway)	Pelagic (and benthic?)	Piatt & Nettleship 1985 Tremblay <i>et al.</i> 2003
<i>Uria aalge</i>	80% dives <50m	138m (Newfoundland) 180m (Newfoundland)	71.7-107.7s (various locations)			Barrett & Furness 1990 Daunt <i>et al.</i> 2003 Burger & Simpson 1986 Camphuysen 2005

Razorbill <i>Alca torda</i>	50% dives <15m (Baltic) <20m during night (Baltic) 25m–30m (Norway) <35m (Iceland)	41m (Iceland) 43m (Baltic) 120m (Newfoundland)	Unknown	Surface diver. V-shaped dives (Baltic); W- and U- shaped dives (Labrador)	Pelagic (and benthic?)	Grémillet <i>et al.</i> 1999 D'all Antonia <i>et al.</i> 2001 Benvenuti <i>et al.</i> 2001 Piatt & Nettleship 1985 Paredes <i>et al.</i> 2008 Barrett & Furness 1990
Atlantic puffin <i>Fratercula arctica</i>	75% dives <10m (laboratory) 25m–30m (Norway)	68m (Newfoundland)	60% <40s (laboratory)	Surface diver	Pelagic	Piatt & Nettleship 1985 Burger & Simpson 1986
Black guillemot <i>Cepphus grylle</i>	10m	20m	43s (Scotland)	Surface diver	Benthic (and pelagic?)	Snow & Perrins 1988
Great cormorant <i>Phalacro- corax carbo</i>	6m (France)	32m (France)	Mean 40s; max. 152s (France)	Surface diver	Pelagic and benthic	Snow & Perrins 1988
European shag <i>Phalacro- corax aristotelis</i>	33m–35m (Scotland) 55% time at 25m–34m (Scotland) 10m–43m (Scotland) Forages most frequently in water depths 21m–40m (Scotland) c. 12–42m	43m (Scotland)	97s (Scotland) 38s 62–71s short range trips 50–58s long- range trips c. 40–110s increased with depth	Surface diver. Almost vertical descent and ascent	Pelagic and benthic	Wanless <i>et al.</i> 1991 Grémillet <i>et al.</i> 1996 Wanless <i>et al.</i> 1997a Wanless <i>et al.</i> 1998 Watanuki <i>et al.</i> 2005 Grémillet <i>et al.</i> 1998a
Northern gannet <i>Morus bassanus</i>	5m (Newfoundland) 20m (Shetland/ Orkney)	22m (Newfoundland) 34m (Shetland/ Orkney)	1s–8s (Shetland) 8s–38s (Shetland/ Orkney)	Plunge diver. Deep, extended U- shaped dives and rapid, shallow V- shaped dives (Shetland/ Orkney)	Pelagic	Garthe <i>et al.</i> 2000 Brierley & Fernandes 2001 Garthe <i>et al.</i> 1999
Northern fulmar <i>Fulmarus glacialis</i>	3m (Shetland)	4m	Max. 8s (Shetland)	Plunge diver	Pelagic	Hobson & Welch 1992 Garthe & Furness 2001
Manx shearwater <i>Puffinus puffinus</i>	14m (wedge- tailed shearwater) 15m (Audubon's shearwater) 21m (black- vented shearwater)	66m (wedge- tailed shearwater) 35m (Audubon's shearwater) 52m (black- vented shearwater)	<20s (Audubon's shearwater)	Plunge diver	Pelagic	Burger 2001 Keitt <i>et al.</i> 2000 Martin & Brooke 1991
Common tern <i>Sterna hirundo</i>	<1m	<1m	Unknown	Plunge diver	Pelagic	Kaiser <i>et al.</i> 2006 Kirkham & Nisbet 1987

Arctic tern <i>Sterna paradisaea</i>	<0.20m	<0.35m	Unknown	Plunge diver	Pelagic	Cramp & Simmons 1985 Kirkham & Nisbet 1987
Sandwich tern <i>Sterna sandvicensis</i>	Unknown	Unknown	Unknown	Plunge diver	Pelagic	Snow & Perrins 1988
Little tern <i>Sterna albifrons</i>	Unknown	Unknown	Unknown	Plunge diver	Pelagic	Snow & Perrins 1988
Roseate tern <i>Sterna dougallii</i>	<1m	<1m	Unknown	Plunge diver	Pelagic	Kirkham & Nisbet 1987
Common scoter <i>Melanitta nigra</i>	3m–20m (England/Wales)	Unknown	Unknown	Surface diver	Benthic	Snow & Perrins 1988
Common eider <i>Somateria mollissima</i>	Unknown	42m (Canada)	Unknown	Surface diver	Benthic	Guillemette <i>et al.</i> 1993
Great northern diver <i>Gavia immer</i>	Unknown	60m (Lake Superior)	Unknown	Surface diver	Pelagic and benthic	Schorger 1947
Black-throated diver <i>Gavia arctica</i>	3m–6m	20m (Baltic)	Mean 45s; max. 120s	Surface diver	Pelagic	Dagys & Zydels 2002
Red-throated diver <i>Gavia stellata</i>	Unknown	21m	Unknown	Surface diver	Pelagic and benthic	Snow & Perrins 1988

3. EXPOSURE TIME POPULATION MODELLING (ETPM) – AN ALTERNATIVE APPROACH TO ASSESSING COLLISION RISK TO DIVING BIRDS FOR TIDAL TURBINES

3.1 Broad concept

A critical element of assessing the potential impact of tidal developments on seabirds will be in relation to the SPA populations of species considered to be at risk. Seabird SPAs in the vicinity of zones suitable for tidal energy devices will typically be breeding colonies, possibly with offshore extensions into the adjacent sea. Thus, the risk of adverse impacts on the integrity of the SPA breeding populations due to a proposed development will need to be assessed, making it valuable to have the potential to base such assessments on site-specific data, or on data that can be applied reliably across a wide range of sites.

As discussed, collecting data on the behaviour of diving birds whilst underwater at a specific site is likely to be extremely difficult. However, observations of diving species made from the surface can provide information on the level of activity (e.g. species density, dive frequency and duration). These data could be used to estimate the size of the ‘at-risk’ population at a

given site, and the level of diving activity, taking into account factors such as tidal state (i.e. high/low flow, ebb, flood, etc.) and basic data regarding the proposed development (e.g. number of turbines, depth range of rotors, area of development and the proportion of the zone judged to present a hazard to diving birds).

The size of the population from which the at-risk subset is drawn could be estimated from the known breeding colonies located within a range of reported mean and maximum foraging distances from the site (Thaxter *et al.* 2012). The sensitivity of the wider population to additional mortality amongst the at-risk population would then be investigated using a population model, parameterised using published data (Maclean *et al.* 2007).

Therefore, it is proposed that the following process for assessing the potential impacts from tidal devices for a given seabird species is followed:

1. Develop a population model, from which thresholds of 'acceptable' additional mortality can be estimated.
2. Estimate exposure time in order to derive collision probabilities per unit period of time which correspond to the mortality thresholds generated by the population model.
3. Consider the associated mortality and collision probability estimates to determine the most likely range and the risk that these could lead to a population level impact at a given scale.

In its current form this process does not incorporate calculation of collision risk explicitly, but rather identifies the collision risk which would be associated with a given level of mortality. However, the structure of the exposure time model is such that it could be re-formulated to permit the inclusion of an empirically derived collision probability should this become possible from future work.

3.2 Exposure time modelling

For the purposes of exposure time modelling, the risk posed by tidal devices is expressed as the number of individuals from a given colony killed as a result of a collision. To estimate this we need estimates of the number of breeding birds within the colony, the length of time each bird within the colony is exposed to the devices and the collision rate (probability of collision) per unit time a bird is exposed. For a colony of N seabirds, the number of deaths due to collision with tidal turbines during a given time period (e.g. month) j , D_j , can be estimated as:

$$D_j = \sum_{i=1}^N (1 - e^{-\alpha T_{ij}}) \quad [1]$$

Where T_{ij} is the length of time bird i from the colony is exposed to the devices in month j , and α is the collision rate per unit time the bird is exposed. This assumes birds encounter turbines at random, so that the probability of a bird not colliding with any turbines is given by the zero term of the Poisson distribution, $e^{-\alpha T_{ij}}$, and the probability of an individual colliding at least once is given by $1 - e^{-\alpha T_{ij}}$. This formulation is similar to that used in the Nicholson-Bailey model of host parasitoid population dynamics (Nicholson & Bailey 1935, Begon *et al.* 2006), with turbines analogous to parasitoids or predators, and seabirds analogous to hosts or prey. Providing collisions are relatively uncommon, then equation [1] simplifies, approximately, to:

$$D_j = N\alpha T_j \quad [2]$$

Where T_j is the average length of time a bird from the colony is exposed to the devices in time period j (e.g. one month), and α is the collision rate per unit time the bird is exposed. Total mortality within a year, D , is obtained by summing the individual estimates of mortality for each time period:

$$D = N\alpha \sum_{j=1}^m T_j \quad [3]$$

Where m is the number of time periods within a year. Or, more simply:

$$D = N\alpha T \quad [4]$$

Where T is the mean total length of time an individual is exposed to the devices within a year:

$$T = \sum_{j=1}^m T_j \quad [5]$$

As discussed above, estimating α represents considerable challenges, due to the difficulty of obtaining site specific dive data, however, the model in equation [4] can be rearranged to estimate the value of α which satisfies particular values of D , as outputted by the population model.

$$\alpha = \frac{D}{NT} \quad [6]$$

This exposure time model makes the simplifying assumption that the collision rate is sufficiently small that it can be assumed to equal the probability of collision. It also assumes that the additional mortality accruing from the turbines is sufficiently small that its effect on the number of animals available to die in subsequent time periods can be ignored (i.e. in Equation [2], $D_j \ll N$ so that N remains constant throughout time period j , and in equations [4] and [6], $D \ll N$ so that N remains constant throughout the year). Both of these assumptions will cause the number of animals dying to be slightly over-estimated. In other words the model is conservative in these respects. Whilst extending the model to incorporate changes in colony size throughout the year would be possible, in practice this would be unlikely to improve the accuracy of estimates, given the approximate nature of colony size estimates and the lack of data on other mortality factors.

3.3 Estimating exposure time

The colony size, N , can be estimated reasonably readily from colony counts. However, estimating the mean individual exposure times, T_j , for each time period is a greater challenge. This section outlines possible approaches to estimating this parameter for both the breeding and non breeding seasons.

3.3.1 Estimating exposure time during the breeding season

During the breeding season, one possible way to estimate T_j from field data would be as follows:

$$T = FPUHS \quad [7]$$

Where:

F_j is the mean number of foraging trips made by an individual within period j .

P_j is the proportion of these foraging trips being made to the development area in period j .

U_j is the mean number of dives on each foraging trip during period j .

H_j is the mean length of time during each dive spent at vulnerable depths (i.e. the same depths as the moving parts of the devices) during period j .

S is the proportion of the water at vulnerable depths, occupied by the parts of the devices with which the birds might collide (e.g. turbine blades).

Similarly, the proportion of foraging trips made to the development area (P) may be estimated either empirically through direct observations across a wide area including the development site, or from a combination of localised observations at the development site in conjunction with colony-based observations or published data on the foraging behaviour of

the target species. The first approach would require boat or aerial surveys across an area encompassing the majority of expected foraging trips from a given colony (or colonies), which depending on the species involved could involve surveys over an extensive area. The proportion of foraging trips made to the site would be calculated as the number recorded on the site divided by the number observed in the whole area surveyed (using the latter as a proxy for the foraging range). For the second approach, the proportion of breeding birds from the colony out at sea at any one time would be estimated from time budget data (using either published sources or data recorded from the colonies in question). Multiplying this figure by the number of individuals at the breeding colony, and then adding the number of non-breeding individuals within this population (as estimated by applying the stable age distribution from the population modelling – see 3.4) would provide an estimate of the total number of birds expected to be foraging at any one time. Dividing the number recorded on the development site by the total number estimated to be foraging therefore yields an estimate of the proportion of birds using the site. In some cases at least, this is likely to represent an overestimate of the proportion of foraging trips made to the development area because it assumes that all birds recorded on site originate from the ‘colony population’ (defined as those birds from both the breeding and non-breeding age classes), which may not be the case.

Estimates of the mean number of dives per foraging trip (U) may also be available from published data, or alternatively by monitoring diving activity on the development site to generate estimates of dive frequency per unit time, which could be used in conjunction with estimates of foraging trip duration (again to be derived from the existing literature, or from further data collection at the relevant colonies).

The mean length of time during each dive spent at vulnerable depths (H) could be estimated using a combination of published data collected using time-at-depth recorders or derived from data collection on dive durations at the development site, which would be used in conjunction with data on water depth and average swim speed estimates (the latter of which is dependent on there being existing published data available).

Finally, the proportion of the water at vulnerable depths that is occupied by the turbine blades and presents a risk (S) could be derived by assuming a simple cylinder with radius equal to the sum of the blade length plus bird body length, and depth equivalent to the rotor width plus twice bird body length (thus accounting for the birds approaching the rotor from any direction). Alternatively, this could be made more sophisticated by recognizing that at any one point in time, blades occupy a proportion of this volume only. For the worked examples given below the simple cylinder estimate was used.

3.3.2 *Estimating exposure time outside the breeding season*

Estimating exposure time outside the breeding season represents a greater challenge than during the breeding season because birds are not constrained to breeding colonies, whilst much of the published information on diving and foraging behaviour of seabirds relates to the breeding season, rather than the non-breeding season. However, provided adequate estimates of dive parameters are available (e.g. Fort *et al.* 2010), it should still be possible to calculate maximum estimates of potential mortality due to collision with the turbines by assuming that all birds within the development area are breeding birds from the colony concerned, and that all the birds from the colony are at sea. However, because the birds remain out at sea rather than return to their breeding colonies, to do so requires a slight reformulation of the terms to calculate the mean time an individual bird is exposed to the turbines in period j (T_j), as follows:

F_j becomes the number of days in period j .

U_j becomes the estimated number of dives per day in period j .

P_j becomes the estimated proportion of birds from the ‘colony population’ (i.e. including the non-breeding age classes) within the development area in period j .
 H_j and S remain unaltered from the breeding season estimation.

A maximum estimate of P can be made by assuming all the birds from the ‘colony population’ are at sea, and all of the birds within the development area (e.g. as determined by counts from boat based surveys) belong to the ‘colony population’. In many cases this will represent a highly precautionary scenario and for some species there may be options to amend the assumption concerning the derivation of birds recorded within the development area based upon knowledge of movements and dispersal patterns during the non-breeding season.

3.3.3 Incorporating variation between subsets of the population

The parameters F_j , P_j , U_j and H_j could potentially vary between subsets of the population, according to such factors as age, sex or stage in the breeding cycle (e.g. incubation or chick rearing). The model in equation 2 can be generalized to include variation between subsets as follows:

$$D_j = \alpha S \sum_{i=1}^L N_{jk} F_{jk} P_{jk} U_{jk} H_{jk} \quad [8]$$

Where:

L is the number of population subsets.

N_{jk} is the number of individuals in subset L during period j .

F_{jk} is the mean number of foraging trips made by an individual of subset k during period j .

P_{jk} is the mean proportion of foraging trips to the development area for individuals of subset k in period j .

U_{jk} is the mean number of dives on each foraging trip for individuals of subset k during period j .

H_{jk} is the mean time on each dive spent at vulnerable depths for individuals of subset k in period j .

Therefore, where sufficiently detailed information already exists, or can be collected as part of the process of populating the model, the modelling can be refined to reflect such sources of variation, and potentially produce more accurate exposure time estimates.

3.4 Population modelling

The population modelling that is carried out to complement the exposure time model may be as simple or as complex as is considered appropriate to the situation in question. In the worked examples set out below stochastic models are used in preference to simpler deterministic models, due to the additional realism they provide (Morris & Doak 2002). Importantly, within the context of the current application of the models, they offer a range of options by which an ‘acceptable’ mortality level may be determined, which may be relevant to situations involving small or declining populations and according to the level of precaution applied. Thus, with stochastic models, impacts may be expressed in terms of the resulting difference in the probability of decline (as in the common guillemot *Uria aalge* example below), as opposed to simply considering whether the additional mortality is estimated to lead to decline. The estimation of confidence intervals (CIs) around the mean values also allows the level of variability in the estimated impact to be assessed.

The availability of demographic data (from which the population model is derived) that are both contemporary and relevant to the geographical area of the population at the development site is of critical importance to the overall approach proposed here, as this determines the reliability of the derived thresholds of ‘acceptable’ mortality. As such, emphasis in the current examples was placed on selecting demographic parameters from

recent monitoring and research. In some cases this meant using demographic parameters derived from a small number of years of data (which will increase uncertainty in the estimates), but this disadvantage was considered to be offset by the value of using parameters that are likely to have greater relevance to the current time.

For the examples given below, stochastic population models were produced using the Rramas package available in the statistical software R (R Development Core Team 2012), with models run in R version 2.14.1. This simulates the growth of a population using a standard matrix-based age-structured approach (Morris & Doak 2002). Thus, a stable age distribution for the population was first produced based upon the estimated demographic parameters for the population, and used in conjunction with the estimate of population size to generate the number of individuals in each age class in the start year. The population projection was then calculated over a 25-year period with both environmental and demographic stochasticity incorporated. Environmental stochasticity was incorporated by sampling at each time step from a normal distribution based upon the mean and standard deviation (SD) of the demographic parameters, with demographic stochasticity incorporated by sampling the number of survivors in each age class at each time step from a binomial distribution and the number of offspring from a poisson distribution. An equal sex ratio was assumed for all modelled populations and the model outputs at each time step represented a post-breeding census (i.e. the number of individuals in the population at the end of the breeding season after all surviving chicks had fledged). No density dependence was incorporated into the models used in the examples below.

For each modelled scenario, 1000 replicates were performed to assess the effects of stochasticity, with 1000 bootstrap samples taken from the simulations to estimate the probability (and associated 95% CI) of declining or increasing to different threshold levels by the end of the projection period. To determine the effect of additional mortality on the population projections, the estimated annual survival rates were recalculated according to the number of additional deaths imposed on the starting population, with the mortality assumed to occur in proportion to the number of individuals within each age-class.

3.5 Worked examples: 1. European shag

The following section details a worked example of the above method for assessing the potential impacts of tidal turbines on European shag *Phalacrocorax aristotelis*, using data available for an unspecified proposed tidal development in the Pentland Firth, northeast Scotland.

3.5.1 Population model

Demographic rates for shag were taken from studies undertaken on the Isle of May, in the outer Firth of Forth, eastern Scotland.

Breeding success estimates were obtained using data from the long-term monitoring of seabird populations undertaken on the Isle of May (<http://www.ceh.ac.uk/sci/programmes/2012-seabird-breeding-IsleofMay.html>). Annual estimates were used for the period 2007 – 2012, giving an estimated mean breeding success (and associated SD) of 0.813 (0.214) fledglings per breeding adult over these years. No recent data were known to be available from other sites within or close to the region of interest.

Age of first breeding was assumed to be three years, with all birds of age 3 years or over assumed to attempt to breed in each year (Frederiksen *et al.* 2008). Estimates of annual survival rates for the different age classes were taken from Frederiksen *et al.* (2008), and were for the period 1963 to 2005. The associated SDs for these estimates were calculated

from the standard errors provided by Frederiksen *et al.* (2008). Thus, the baseline mean annual survival rates (with SDs) used in the final population model were as follows:

- Breeding adults – 0.858 (0.194)
- 1–2 years – 0.737 (0.181)
- Juveniles – 0.513 (0.246)

No population trend data were known to be available from the region around the Pentland Firth from which to assess the likely reliability of the modelled population trend.

The initial population size was set at 1181 individuals, as derived from an estimated adult breeding population of 606 within a 20km radius of the site (Mitchell *et al.* 2004) considered in conjunction with the stable age distribution predicted from the population model (i.e. juvenile:immature:adult proportions of 0.33:0.16:0.51). The 20km range is based on the estimated maximum foraging range for this species (Langston 2010). With no additional mortality imposed upon the population (i.e. using the baseline annual survival rates above), the model predicted a mean population growth rate (λ) of 1.06, producing an estimated mean population size of 5823 (± 12955 SD) after 25 years. There was a high degree of uncertainty around the predicted population sizes, particularly as the period of the prediction increased (Figure 1). This was due to the large variability in the estimated annual survival rates (as indicated by the associated SDs), reflecting in part the effects of extreme weather on the survival of certain age classes (Frederiksen *et al.* 2008). The high level of uncertainty in the modelled projection means that any conclusions concerning the assessments of potential impacts on the population have to be treated with caution.

Given that the baseline prediction was for an increasing population, for the purposes of this exercise, levels of impact were considered in terms of whether they were predicted to be sufficient to produce a population decline, or at least stability (as opposed to considering changes in the probability of decline). Thus, in this case, additional mortality was considered to be 'acceptable' if it did not prevent increase in the predicted mean population size. Additional mortality was imposed in increments equivalent to removing 20 individuals (distributed between age classes in proportion to the stable age distribution) from the starting population until the population was no longer predicted to increase under an additional mortality of 60 individuals. With this level of additional mortality, the mean population size at 25 years was predicted to be 1172 (± 2229 SD) individuals ($\lambda = 1.00$).

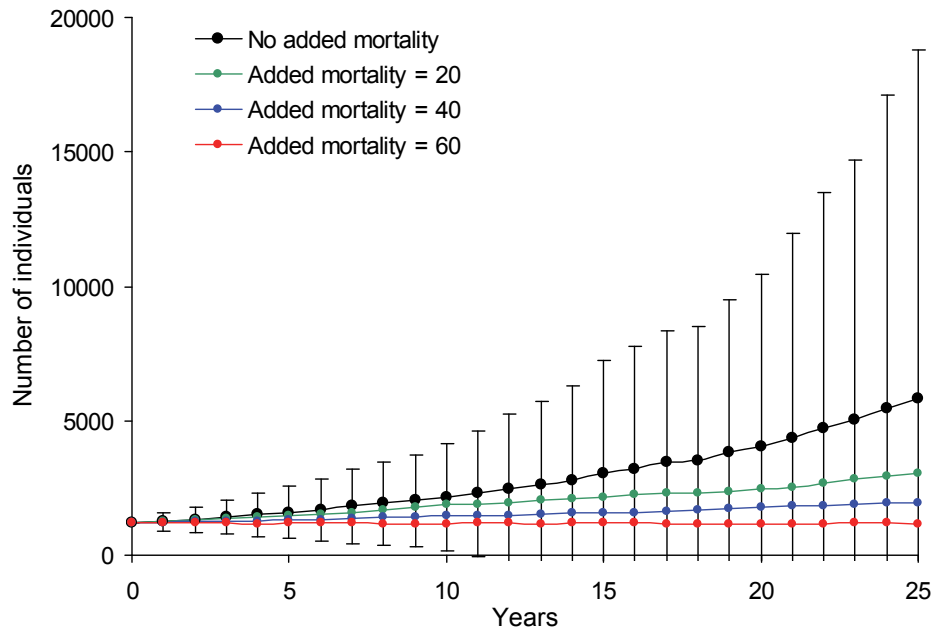


Figure 1. The projected population growth for the example European shag population in the absence of any additional mortality, and with different levels of additional mortality imposed. The predicted mean population sizes ($\pm 1SD$) are shown for the baseline scenario but (to aid presentation) predictions incorporating additional mortality show the mean values only. The added mortality represents the number of additional deaths imposed on the starting population for each scenario, with survival rates reduced proportionately (see text).

3.5.2 Exposure time model

Exposure time for the example shag population during a 12 month period, comprising three months of breeding season (May – July) and nine months of non-breeding was estimated using equation [7] (Table 2). Thus, during the breeding season, each individual's exposure time (T) per month was calculated as a function of the total number of foraging trips (F), the proportion of those trips made to the development site (P), the mean number of dives per trip (U), the mean duration spent at vulnerable depths (H) and the proportion of the water at the vulnerable depth occupied by turbine blades (S). During the non-breeding season F , P and U were defined as; (i) the number of days per period (e.g. month); (ii) the proportion of the 'colony population' (inclusive of immatures and juveniles) on the site; and (iii) the mean number of dives per day, respectively. Details of the derivation of each of the components of exposure time (T) are described below.

F – total number of individual foraging trips: Wanless *et al.* (1997b) estimated that during incubation breeding adults on the Isle of May undertook one foraging trip per day whilst during chick rearing they made three trips per day. Analysis of food loads of shag provisioning young on the Chausey Islands, Normandy, indicated an average of 3.39 foraging trips per adult per day (Grémillet *et al.* 1996). Therefore, it was assumed that each adult made one trip per day during incubation (May), and three trips per day during June and July (chick rearing). Outside the breeding season (and for non-breeding age classes at all times) it was assumed that each bird made one foraging trip per day. For all months the value of F was obtained by multiplying the number of daily trips by the number of days in the month.

P – proportion of foraging trips made to development site: Shags only forage during daylight, whilst during incubation and chick rearing one adult always remains at the nest, so the

percentage of breeding birds actively foraging at any one time cannot exceed 50%. Therefore, with day lengths of up to 16-18 hours in northern Scotland, each individual could potentially spend 8-9 hours foraging. Assuming three foraging trips per day means that each trip could take up to three hours. However, studies on the Isle of May and the Chausey Islands have estimated maximum trip durations in the region of 90 and 120 minutes, respectively (Grémillet *et al.* 1996; Wanless *et al.* 1998). Thus, the proportion of breeding adults assumed to be actively foraging at any one time was reduced from 0.5 to 0.33, based on an average foraging trip duration of two hours. The number of breeding adults within 20km of the site (the maximum foraging range as defined by Langston 2010) was estimated to be 606 using data from Mitchell *et al.* (2004), which equated to a total population of 1181 individuals (based upon the stable age distribution from the population model which estimated that breeding adults comprised 51% of the population). Therefore, 775 birds from the population were calculated to be foraging at any one time during the breeding season (i.e. 202 breeding adults plus all immatures), and all 1181 birds at any one time outside the breeding season. Boat surveys of the site were undertaken to record monthly densities for birds on the water, from which estimates of the proportion of the population present on the site in each month were derived.

U – mean number of dives per trip: Studies on the Isle of May and the Chausey Islands indicated that shags undertook an average of 49 and 58 dives per trip, respectively, during the breeding season (Wanless *et al.* 1998, Grémillet *et al.* 1998b). The average of these values was used for both the breeding and non-breeding seasons (no data being found for the number of dives per day outside the breeding season).

H – mean duration at vulnerable depths: The mean water depth at the development site is 30m. Based upon current proposals it is assumed that the rotor for the tidal turbine will not be contained within a cowl and that it would have a diameter of 20m. Shag are typically considered to forage on or just above the seabed (Harris & Wanless 1991, Wanless *et al.* 1991b) therefore the birds are only at vulnerable depths whilst travelling to and from the bottom. Based upon a swim speed of 1.65m/s (equivalent to the mean of the mid-points of the minimum and maximum descent and ascent swim speeds – Watanuki *et al.* 2005), then an individual bird passes through the rotor swept portion of the water column in 12.1s, reaching the seabed in 18.2s, and giving a value for *H* of 24.2 seconds.

S – proportion of water at vulnerable depths containing rotors: The estimate of the proportion of water at vulnerable depths occupied by the rotors was based on a hypothesised example development comprising 100 turbines within a seabed area of 1km². Each individual turbine was assumed to comprise a single rotor of 20m diameter, with blades 0.5m deep. The body length of a shag is estimated as 0.75m (Snow & Perrins 1998), so that the volume of water presenting a potential risk to a diving shag (*V*) would be:

$$V = 100 * ((\pi(10 + 0.75)^2) * (0.5 + 1.5)) = 72610\text{m}^3 \quad [9]$$

And the proportion of water at vulnerable depths occupied by 100 turbines, *S*:

$$S = 72610 / ((20 + (2*0.75)) * 10002) = 0.00338 \quad [10]$$

Table 2. Calculation of exposure time parameter estimates for an example European shag population in the Pentland Firth, based on a combination of observational and published data and a tidal turbine installation covering 1km².

Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
No. of foraging trips (<i>F</i>)*	31	28	31	30	31	90	93	31	30	31	30	31
No. individuals on site (95% C.I.)#	14.0 (10.0-19.6)	9.9 (7.2-13.8)	18.0 (10.7-30.0)	7.5 (5.7-9.8)	10.0 (6.4-15.6)	6.4 (3.8-10.8)	14.1 (8.7-22.7)	12.3 (9.1-16.5)	13.9 (7.3-26.4)	3.6 (1.8-7.3)	12.4 (6.5-23.4)	2.2 (1.1-4.4)
Potential no. of foraging individuals in source population	1181	1181	1181	1181	775	775	775	1181	1181	1181	1181	1181
Proportion of foraging trips made to the site (<i>P</i>)	0.012	0.008	0.015	0.006	0.013	0.008	0.018	0.010	0.012	0.003	0.010	0.002
Mean no. of dives per trip (<i>U</i>)						53.5						
Mean duration (s) at vulnerable depths during foraging dive (<i>H</i>)						24.2						
Proportion of water at vulnerable depths containing rotors (<i>S</i>)						0.00338						
Monthly exposure time (s) (<i>T</i>)	1.61	1.03	2.06	0.83	1.75	3.23	7.39	1.41	1.54	0.41	1.37	0.25
Annual exposure time (s) (<i>T</i>)						22.90						

*for breeding season months (May - July) refers to the breeding age class

#derived by applying distance sampling to transect counts from boat surveys (Buckland *et al.* 2001).

From the estimation of the above components of exposure time, monthly values for exposure time are calculated and summed to give an estimated annual value of 22.9 s for the example shag population (Table 2). Therefore, each of the 1181 individuals in the population is at risk of collision for 22.9s per year. Substituting the annual value for *T* and the range of additional mortality estimates from the population modelling (Figure 1) into equation

[6] generates the collision probability required to account for the additional mortality (Table 3). As described earlier (section 3.2), collision probability is assumed to be equivalent to collision rate, and therefore at least 0.0022 collisions per second are estimated to be required for the additional mortality to be sufficient to cause a decline in the shag population (i.e. an additional 60 deaths in the starting population, equivalent to increasing annual mortality by 5.08% - Table 3). This approximates to one collision for every 7 - 8 minutes that birds spend within the rotor swept water volume. This low value suggests that additional mortality resulting from collisions with turbines could potentially have a serious impact on the associated breeding population, and would need to be considered in any impact assessment.

Table 3. Levels of collision probability required to account for the different additional mortalities applied in the European shag population models for the given estimated exposure time.

No. of individuals in source population (N)	Annual exposure time (s) (T)	Initial additional mortality (percentage increase in annual mortality) (D)	Collision probability (α)
1181	22.9	20 (1.69)	0.00074
		40 (3.39)	0.00148
		60 (5.08)	0.00222

3.6 Worked examples: 2. Common guillemot

The second example for assessing the potential impacts of tidal turbines using the ETPM approach is for common guillemot, and as above it uses data available for an unspecified proposed tidal development in the Pentland Firth, Scotland.

3.6.1 Population model

As for shag, demographic rates for guillemot were derived from studies undertaken on the Isle of May.

Thus, annual estimates of the number of chicks produced per nest on the Isle of May were used for the period 2007 - 2012 (http://www.ceh.ac.uk/sci_programmes/2012-seabird-breeding-IsleofMay.html), giving an estimated mean breeding success (and associated SD) of 0.330 (0.098) fledglings per breeding adult over those years. No recent data were known to be available from other sites within or close to the region of interest.

Age of first breeding was taken as 5 years, and it was assumed that all birds of age 5 years or over attempted to breed in each year (Harris *et al.* 2007). Reynolds *et al.* (2010) provide estimates of annual survival rates of guillemots from the Isle of May for the period 1983 – 2003, with the estimates for the different age classes (along with their SDs) being used to produce the initial population model for the present work. However, this initial model predicted an increasing population, whereas the available evidence on guillemot population trends in the region around the Pentland Firth indicate possible declines in breeding numbers, at least in the large colonies on Hoy and at Marwick Head in Orkney (Lewis & Lye 2012). The demographic data from the Isle of May suggest a decline in annual juvenile survival rates during the most recent period for which estimates are available (Reynolds *et al.* 2010), and therefore the value for mean juvenile survival provided by Reynolds *et al.* (2010) was replaced in the population model by the mean of the values from 1999 - 2003,

inclusive (as derived by reading off Figure 2 in Reynolds *et al.* 2010). The SD for the mean of these years was also used.

The baseline mean annual survival rates (with SDs) used in the final population model were as follows:

- Breeding adults – 0.910 (0.018)
- 3 - 4 years – 0.930 (0.017)
- 1 – 3 years (composite value) – 0.760 (0.024)
- Juveniles – 0.360 (0.190)

The initial population size was set at 106447 individuals, as derived from an estimated adult breeding population of 71000 within 61km of the site (Mitchell *et al.* 2004), considered in conjunction with the stable age distribution predicted from the population model (i.e. juvenile:immature:adult proportions of – 0.20:0.13:0.67). The range is based on the estimated mean maximum foraging range for this species (Langston 2010). With no additional mortality imposed on the population (i.e. using the baseline annual survival rates above), the predicted mean population growth rate was 0.99, producing an estimated mean population size of 81575 (± 32885 SD) after 25 years (Figure 2).

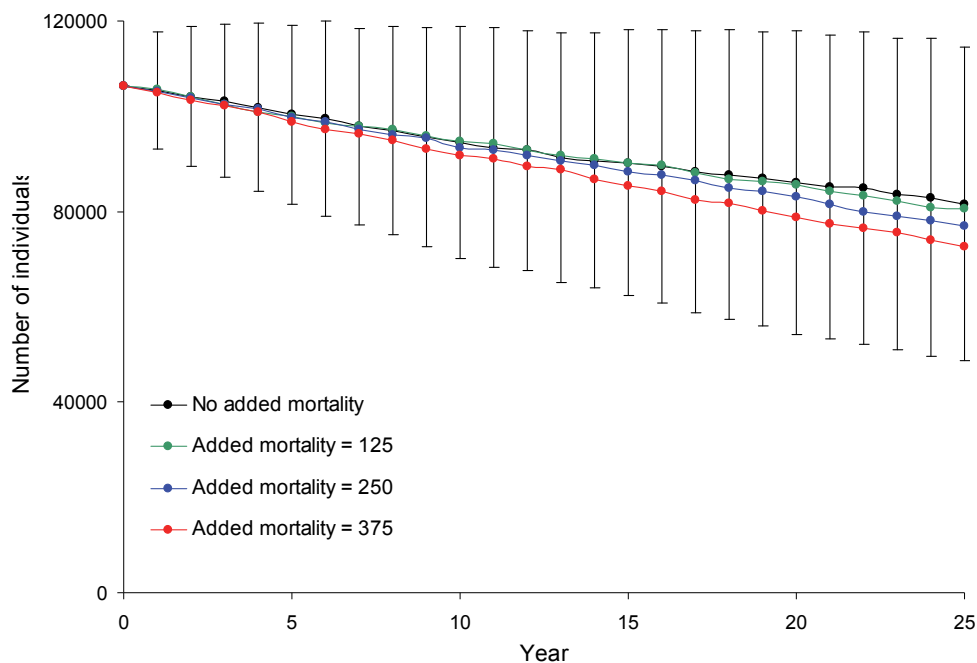


Figure 2. The projected population growth for the example Common guillemot population in the absence of any additional mortality, and with different levels of additional mortality imposed. The predicted mean population sizes ($\pm 1SD$) are shown for the baseline scenario but (to aid presentation) predictions incorporating additional mortality show the mean values only. The added mortality represents the number of additional deaths imposed on the starting population for each scenario, with survival rates reduced proportionately (see text).

Determining what might be considered an ‘acceptable’ level of additional mortality is potentially more problematic in this example than in the previous example for the shag population, because the baseline prediction in this case is for a declining, as opposed to increasing, population. For the purposes of this exercise, the assessment of an ‘acceptable’ level of additional mortality was undertaken by examining increases in the probability that declines of particular magnitudes would occur, focussing on declines of at least 25% and

50% over the 25 year projection period. In the absence of any additional mortality, such declines were predicted to have a 77% and 28% chance, respectively, of occurring (Table 4). Additional mortality was imposed in increments equivalent to removing 125 individuals (distributed between age classes in proportion to the stable age distribution) from the starting population until there was a statistically significant increase in the probability of the population declining by either 25% or 50% (as determined from non-overlapping 95% CIs). For both thresholds of decline, such an increase in probability was predicted to occur under an additional mortality of 375 individuals, although a mortality level of 250 individuals produced a near significant increase for the 50% decline threshold (Table 4). With an additional mortality of 375 individuals, the mean population size at 25 years was predicted to be 72733 (± 28367 SD) individuals, $\lambda = 0.98$ (Figure 2).

Table 4. The predicted mean (with 95% C.I.) probability of the example common guillemot population declining by at least 25% and 50%, with increasing levels of additional mortality imposed.

Threshold of decline	Level of initial additional mortality (% increase in annual mortality)			
	0 (0)	125 (0.12)	250 (0.23)	375 (0.35)
25%	0.77 (0.75 – 0.80)	0.79 (0.77 – 0.82)	0.80 (0.78 – 0.83)	0.84 (0.81 – 0.86)
50%	0.28 (0.25 – 0.30)	0.27 (0.24 – 0.30)	0.33 (0.30 – 0.35)	0.37 (0.34 – 0.40)

3.6.2 Exposure time model

Exposure time for the example guillemot population during a 12 month period, comprising two months of breeding season (May – July) and nine months of non-breeding was estimated using equation [7] (Table 5). Thus, during the breeding and non-breeding seasons, each individual's exposure time (T) per month was calculated in the same way as for the example shag population (see 3.5 above). Details of the derivation of each of the components of exposure time (T) are described below.

F – total number of individual foraging trips: Thaxter *et al.* (2009) estimated that whilst provisioning chicks, breeding females and males on the Isle of May undertook 2.8 and 2.3 trips per day respectively. It was assumed that this level of foraging occurred during June (giving a mean of 2.55 foraging trips per breeding adult), but that during incubation (nominally May), only one trip was made per day. A single foraging trip per bird per day was also assumed for July, when most breeding males will have left the colony and be at sea with dependent chicks (chicks leaving the nest at an average age of 22 days and when still flightless), but when breeding females will continue to visit the nest-site (Wanless *et al.* 2005).

Outside the breeding season (and for non-breeding age classes at all times) it was assumed that each bird made one foraging trip per day. For all months the value of F was obtained by multiplying the number of daily trips by the number of days in the month.

P – proportion of foraging trips made to development site: During the incubation and chick rearing stages at the colony (i.e. nominally May and June) one adult always remains at the nest, so the percentage of breeding birds actively foraging at any one time cannot exceed 50%. Trip duration data suggest that on average breeding adults spend 46.5% of their time

away from the nest site (Thaxter *et al.* 2009). The number of breeding adults within 61km of the site (the mean maximum foraging range as defined by Langston 2010) was estimated to be 71000 (Mitchell *et al.* 2004), which equated to a total population of 106447 individuals (based upon the stable age distribution from the population model). Therefore, 68462 birds from the population were calculated to be foraging at any one time during May and June (i.e. 33015 breeding adults plus all immatures), and all 106447 birds at any one time in other months (which is acknowledged to overestimate foraging numbers in July because females continue to visit nest-sites at this time). Boat surveys of the site were undertaken to record monthly densities for birds on the water, from which estimates of the proportion of the population present on the site in each month were derived.

U – mean number of dives per trip: Studies on the Isle of May indicated that guillemots undertook an average of 78.4 dives per trip (Thaxter *et al.* 2009). No data were found for the number of dives made per day outside the breeding season, therefore the same value was adopted for all months.

H – mean duration at vulnerable depths: As detailed in the example for shag, the mean water depth at the development site is 30m, whilst a 20m turbine rotor diameter is considered likely. Guillemots typically have ‘U’ shaped dive profiles, and feed either at the bottom or in horizontal prey layers (Watanuki *et al.* 2006). The extent to which guillemots at this site feed on the sea-bed is unknown, and so the conservative assumption was made that all foraging occurred within rotor swept depths. Based upon observations of feeding guillemots at this site, the average dive duration was estimated as 34.9s (95% CI: 24.9 – 44.9; n=31). Assuming an average swim speed of 1.69 m/s (i.e. the mean of the descent and ascent swim speeds – Watanuki *et al.* 2006), means that individuals were estimated to spend 25.5s at rotor depth on each dive.

S – proportion of water at vulnerable depths containing rotors: As in the example for shag, a hypothesised development comprising 100 turbines within a seabed area of 1km² was assumed. Thus, details of the equations for estimating S are as for the shag example, except that the body length for a guillemot was taken to be 0.41m (Snow & Perrins 1998), so that the volume of water presenting a potential risk to a diving guillemot (V) would be:

$$V = 100 * ((\pi (10 + 0.41)^2) * (0.5 + 0.82)) = 44939\text{m}^3$$

And the proportion of water at vulnerable depths occupied by 100 rotors, S:

$$S = 44939 / ((20 + (2*0.41)) * 10002) = 0.00216$$

Table 5. Calculation of exposure time parameter estimates for an example Common guillemot population in the Pentland Firth, based on a combination of observational and published data and a tidal turbine installation covering 1km².

Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
No. of foraging trips (F)*	31	28	31	30	31	76.5	31	31	30	31	30	31
No. individuals on site (95% C.I.)#	0.0 (0.0-0.0)	0.8 (0.3-1.9)	1.8 (0.9-3.6)	1.5 (0.8-2.9)	5.1 (3.7-7.1)	8.5 (5.2-13.9)	23.9 (11.9-48.2)	0.0 (0.0-0.0)	1.2 (0.5-2.6)	0.5 (0.1-1.9)	0.5 (0.2-1.4)	0.1 (0.0-0.5)
Potential no. of foraging individuals in source population	106447	106447	106447	106447	68462	68462	106447	106447	106447	106447	106447	106447
Proportion of foraging trips made to the site (P) *10 ⁻⁵	0.00	0.72	1.64	1.43	7.45	12.37	22.47	0.00	1.08	0.47	0.50	0.07
Mean no. of dives per trip (U)							78.4					
Mean duration (s) at vulnerable depths during foraging dive (H)							25.5					
Proportion of water at vulnerable depths containing rotors (S)							0.00216					
Monthly exposure time (s) (T) *10 ⁻³	0.00	0.87	2.20	1.85	9.95	40.78	30.01	0.00	1.40	0.63	0.64	0.10
Annual exposure time (s) (T)							0.0884					

*for the main chick rearing month at the colony (i.e. June), values refer to the breeding age class

#derived by applying distance sampling to transect counts from boat surveys (Buckland *et al.* 2001).

Note, values of P are given as *10⁻⁵ and of T as *10⁻³

From the estimation of the above components of exposure time, monthly values for exposure time are calculated and summed to give an estimated annual value of 0.09s for the example guillemot population (Table 5). Therefore, each of the 106447 individuals in the population is at risk of collision for 0.09s per year. Substituting the annual value for T and the range of additional mortality estimates from the population modelling (Figure 2, Table 4) into equation [6] generates the collision probability required to account for the additional mortality (Table 6). As described earlier (section 3.2), collision probability is assumed to be equivalent to collision rate, and therefore 0.04 collisions per second are estimated to be required to cause sufficient additional mortality for a statistically significant increase in the probability of the guillemot population declining by either 25% or 50% over 25 years (i.e. an additional 375 deaths in the starting population, equivalent to increasing the annual mortality by 0.35% -

Table 6). However, a rate of 0.0267 collisions per second gives an additional mortality of 250 individuals (i.e. an increase of 0.23% in the annual mortality rate), which produced an increase in the probability of population decline that bordered on statistical significance for the 50% threshold (Table 5). Thus, 'acceptable' levels of additional mortality (as defined for the purposes of this example) may be exceeded by collision rates that are in excess of 0.0267 - 0.04 per second, equating to approximately one collision every 25 - 40s that birds spend within the rotor swept water volume.

This value is substantially higher than that estimated in the earlier example for shag, and suggests that serious impacts of collision mortality on the guillemot population are considerably less likely than for the shag population, as would be expected from the relatively large 'source' population but low densities on the development site for guillemot. However, when considered in relation to typical rotation speeds for single rotor turbines of 10-20 rpm, it is evident that collision rates of one per 25-40s are feasible. Against this, it has to be considered that some avoidance and evasive behaviour can be assumed to occur, reducing the likelihood of collisions. Thus, the level of collision mortality estimated to be required to cause an 'impact' on the population is not so high that it can be entirely discounted, although it seems unlikely to occur.

Table 6. Levels of collision probability required to account for the different additional mortalities applied in the Common guillemot population models for the given estimated exposure time.

No. of individuals in source population (N)	Annual exposure time (s) (T)	Initial additional mortality (percentage increase in annual mortality) (D)	Collision probability (α)
106447	0.0884	125 (0.12)	0.0133
		250 (0.23)	0.0267
		375 (0.35)	0.0400

4. SENSITIVITY ANALYSIS

The annual exposure time, T , is derived from a number of parameters, several of which have uncertainty associated with their estimation (e.g. bird density on the development site and bird swim speeds), whilst others, although absolute measurements, may be subject to potential variation according to deployment on the development site (e.g. diameter of rotor blades on turbines and the number of turbines). Therefore, a sensitivity analysis was undertaken to assess, firstly, how adjustments to values of the different parameters may affect the predicted vulnerability of diving birds and, secondly, how variation within the likely level of error for certain parameters affects the estimation of T (so indicating the level of precision with which these parameters need to be measured for the reliable estimation of T). The effect of variation in T on the collision rate estimated to be required to exceed 'acceptable' levels of additional mortality is also examined. The sensitivity analysis uses data from the shag and guillemot examples above to illustrate the likely impact of variation in parameter values.

4.1 Effects of varying parameter values

To assess the sensitivity of T to variation in different parameters, values for each parameter were varied by 5%, 10% and 20% while maintaining all other parameters at the values used in the actual exposure time modelling examples above.

Being a simple product of five components (i.e. F , P , U , H and S), annual exposure time (T) changes in direct proportion to changes in each of these components (Figure 3). However, these components are themselves estimated from a range of different input parameters so that effects of varying the values of the following input parameters were also tested:

- Number of foraging trips per day
- Number of individuals on site
- Potential source population (i.e. in terms of the potential number of foraging individuals)
- Bird swim speed
- Bird body length
- Dive duration (guillemot example only)
- Number of turbines
- Rotor diameter
- Rotor blade depth

Of these parameters, three are directly proportional to the component of relevance, and hence to T (i.e. the number of foraging trips, the number of individuals on site and turbine number). Other parameters represent (or contribute to) the denominator for calculating one of the components of T (i.e. the potential source population, bird body length and, for the shag example, bird swim speed). Thus, for these parameters, changes to the values do not produce a directly proportional change in T , whilst in some cases the relationships are also inverse (Figure 3). However, in each of these cases the sensitivity of T to changes in parameter values does not differ greatly in terms of magnitude from that associated with changes in those parameters that are directly proportional, at least over the range of variation examined. Thus, a 20% increase in the value for the potential source population (and in the shag example for swim speed also) causes a 25% reduction in T , with 10% and 5% increases causing 11% and 5% reductions in T , respectively. Whilst 20%, 10% and 5% decreases in the potential source population cause 17%, 9% and 5% increases in T , respectively. Sensitivity to changes in bird body length is closer to a directly proportional effect (Figure 3).

Methods for calculating the time spent at vulnerable depths (H) differed between the shag and guillemot examples, due to differences in the likely foraging behaviour of these species (with shags assumed to forage on the seabed and guillemots assumed to undertake all foraging at rotor swept depths - see 3.5 and 3.6 above). Thus, the sensitivity of T to variation in swim speed is less marked (and the relationship direct and not inverse) in the guillemot example (Figure 4). In the guillemot example, T is also relatively sensitive to changes in dive duration (a parameter not used in calculating T for shags), which cause a 1.37 fold change in T .

For both of the examples, T shows greatest sensitivity to changes in rotor diameter, which is a parameter that contributes to the calculation of two components of T (i.e. H and S). Thus, 5%, 10% and 20% increases in diameter cause 10%, 20% and 42% increases in T , respectively, with equivalent decreases in diameter causing 9%, 18% and 35% reductions in T , respectively. Conversely, T is least sensitive to changes in the depth of the rotor blades (Figure 3), which is a constituent of one of several factors that comprise the numerator in the calculation of S .

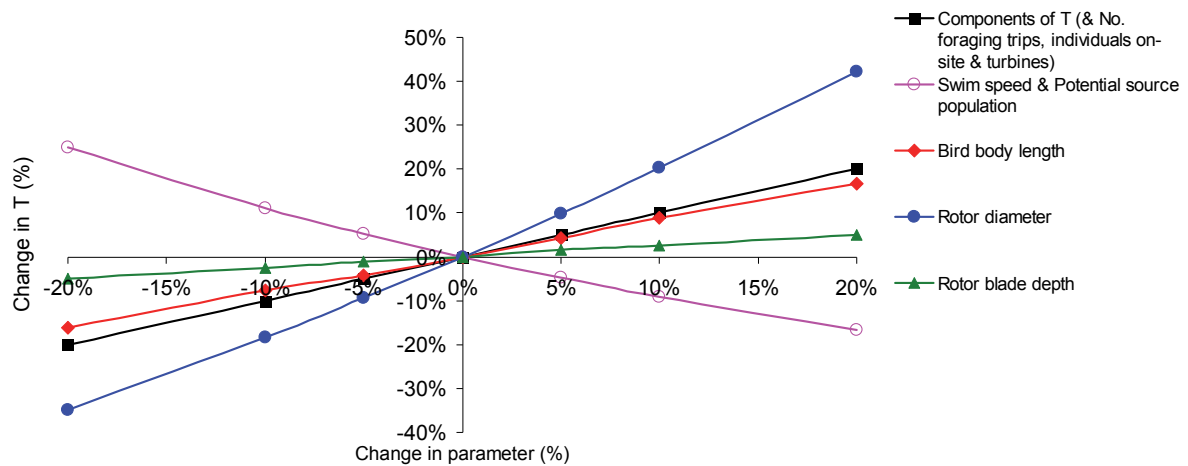


Figure 3. The effect of varying the values of a range of input parameters on the estimation of annual exposure time (T), based upon the calculation of T in the example for European shag. The 'Components of T ' (represented by the black squares and line) are defined as the five attributes (F , P , U , H and S) of which T is the product (see text).

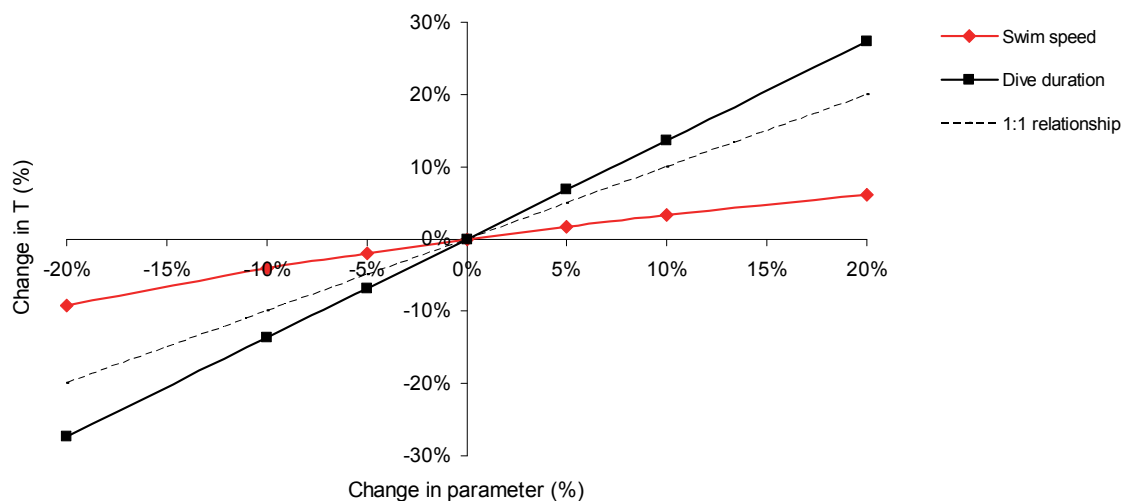


Figure 4. The effect of varying the values of bird swim speed and dive duration on the estimation of annual exposure time (T), based upon the calculation of T in the example for common guillemot. The effects of variation in the values of other input parameters in the guillemot example are as shown in Figure 3 above. The 1:1 relationship (equivalent to that for the 'Components of T ' in Figure 3) is shown for comparison.

4.2 Effects of uncertainty in the estimation of parameters and implications for deriving the 'required' collision rate

As indicated above, several of the parameters that contribute to the estimation of T have uncertainty associated with their estimation. For some of these parameters, the variation about the estimate has been measured, whilst for others alternative values of the parameter are available. Such information is used to extend the sensitivity analysis and explore the level of error that may result in the estimation of T as a consequence of uncertainty in the estimation of certain input parameters. This is undertaken for the estimates of the number of

individuals on site (for which the 95% CIs have been calculated) for both the shag and guillemot examples and for swim speed and dive rate for the shag example (for which a range of estimates are available) and for dive duration for the guillemot example (for which 95% CIs have been calculated, whilst there are also a range of alternative estimates available from other sources). The subsequent effect of such potential error in the estimation of T on the collision rate that is estimated to be required to exceed the 'acceptable' level of additional mortality (as determined by the population modelling) is also assessed.

From these comparisons, it is apparent that uncertainty in the estimation of certain input parameters may have a marked effect on the value of T , with the estimates in both examples being c.40% lower and c.60–80% higher when based upon the lower and upper 95% CLs, respectively, for the number of individuals estimated to be on site (Table 7, Figure 5). The most marked effect on T resulted from substituting the mean dive duration recorded on the site for guillemots with the maximum value found from other sources, which at 107.7s was three times greater and produced a near 300% increase in T (Table 7, Figure 5). Such a large degree of error in the estimation of T is unlikely because the substituted value derives from an entirely different site, where guillemot foraging behaviour may differ from that on the development site. However, few data are available on annual variation in dive duration on different sites and this at least serves to illustrate the potential level of uncertainty that may exist in the estimation of T as a consequence of the paucity of data on some key parameters. Also, in this example the mean dive duration recorded on the development site for guillemots was markedly lower than that recorded in studies elsewhere (Wanless *et al.* 1988, Camphuysen 2005, Evans *et al.* 2013), suggesting that the foraging behaviour of this species on this site may be atypical.

Table 7. Variation in estimated annual exposure time (T) associated with measured, or potential, levels of variation in the estimates of certain input parameters used in calculating T for the European shag and common guillemot examples. The collision rate (α) required to exceed the 'acceptable' level of additional mortality (i.e. 60 and 375 individuals for shag and guillemot, respectively) for the different values of T is also given (see text for details).

Input parameter	Value	European shag		Common guillemot	
		T	α	T	α
All parameters	As estimated in Tables 2 (shag) and 5 (guillemot)	22.90	0.00222	0.088	0.0400
No. of individuals on site ¹	95% LCL	14.29	0.00356	0.050	0.0705
	95%UCL	37.13	0.00137	0.159	0.0222
Swim speed ²	Lower value	29.07	0.00175	-	-
	Upper value	18.89	0.00269	-	-
Dive rate ³	Lower value	20.96	0.00242	-	-
	Upper value	24.83	0.00205	-	-
Dive duration ⁴	95% LCL	-	-	0.053	0.0665
	95%UCL	-	-	0.123	0.0286
	Lower value	-	-	0.088	0.0400
	Upper value	-	-	0.341	0.0103

¹95% CIs are presented in Tables 2 and 5.

² T is estimated using the mean of the midpoints of the minimum and maximum descent and ascent swim speeds (Watanuki *et al.* 2005), so that a lower value was derived from the mean of the minimum descent and ascent speeds, and an upper value using the mean of the maximum descent and ascent speeds.

³ T is estimated using the mean of two separate estimates of dive rate (Grémillet *et al.* 1998b, Wanless *et al.* 1998), with the individual estimates used to provide lower and upper values.

⁴ T is estimated using the mean dive duration recorded during observations on site, with the 95% CI presented in the text (3.6.2). Lower and upper values are also derived using dive duration data from a range of sites (see text), none of which had a lower values than that recorded on the current site (hence why the lower value is as used in estimating T).

Uncertainty in the estimation of T causes associated uncertainty in the estimation of the collision rate that is required to exceed the level of ‘acceptable’ additional mortality, as identified from the population models (Figures 1 and 2). For example, the level of variation encompassed within the 95% CIs for the number of individuals on the site gives a range in the required collision rate of 0.00137–0.00356 for shag and 0.0222–0.0705 for guillemot, giving values that range from approximately 55–60% less than the mean to 60–75% greater than the mean. The effect of variation in the other parameters examined was less than this, except for the range in dive duration values obtained for guillemot from other sites (and discussed above).

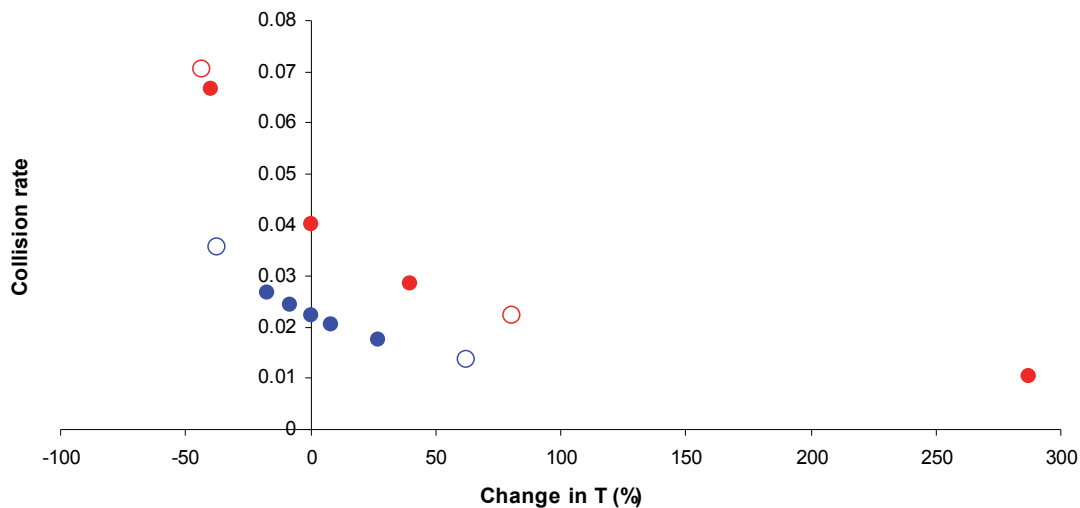


Figure 5. The effect of variation in annual exposure time (T) on the collision rate estimated to be required to exceed the level of ‘acceptable’ additional mortality (as determined from population models) for the Common guillemot (red symbols) and European shag (blue symbols) examples. The variation in T is associated with uncertainty in the estimation of some of the parameters from which T is calculated, with open symbols representing the 95% CIs for estimates of the number of birds on site and closed symbols representing variation in swim speeds and/or dive behaviour (see Table 7 for details). Collision rate values for shag are shown as $\times 10$ for presentational purposes.

5. DISCUSSION

5.1 Uncertainty and the potential for refinements

It is currently extremely difficult to predict the potential impact of tidal turbines on wildlife, since very few devices have been installed. With regards to diving seabirds there are parallels with offshore wind farms for which collision risk estimation has become an established component of impact assessments. The challenge for transferring collision risk modelling to the underwater environment lies chiefly in the difficulty of obtaining detailed, applicable, data on seabird diving parameters, and in understanding behavioural responses

to underwater devices. Modelling the risk to diving seabirds using the individual exposure time approach outlined here sets out the pertinent parameters in a framework that allows currently available data to be used to explore the potential impacts. As well as requiring prediction of the individual exposure time, this approach also relies upon predictions of additional mortality thresholds using population models, although there is no explicit linkage between these two components within the modelling process. Both of these components are key to the prediction of likely effects and are combined to provide a broad-based assessment of the potential risk posed to the population from collision.

Among the assumptions made in the two examples of the modelling approach that are presented here, some of the most important concern the use of generic data measured at other locations, both for aspects of the exposure time modelling (particularly in terms of foraging behaviour) and for the population modelling. As well as being important considerations in assessing the reliability of predictions, they are also aspects where there may be potential to modify and improve input data.

The sensitivity analyses undertaken for this work demonstrate that variation in foraging-related parameters can have marked effects on estimates of T , and highlight the potentially high level of error that can arise from using data derived from other sites. It is also apparent that in many cases there are few estimates of the relevant parameters available from the existing literature (e.g. one and two estimates were found for the number of dives per foraging trip for guillemot and shag, respectively), and where multiple estimates are available, differences can be large (e.g. the two estimates of shag dive frequency differ by almost 20%). The extent to which such differences represent real effects or sampling error is unclear, but it may be expected that foraging behaviour at sites suitable for tidal turbines will differ to some extent from that at other sites because of the different conditions that occur in areas with high tidal flows.

Although there are likely to be benefits in deriving site-specific estimates of foraging behaviour, the extent to which this is feasible will be limited by logistical constraints and available resources. Within the scope of the ornithological work that would generally be undertaken for a specific development, certain types of measurements are more likely to be feasible than others. For example, it may be possible to obtain measurements of attributes such as dive duration and dive frequencies from observational studies that are undertaken through an extension of more standard surveys, without requiring substantially greater resource investment. However, measuring other attributes such as swim speeds and dive depths is considerably more challenging, both in terms of the resource investment required (e.g. due to the need to use devices such as time-at-depth gauges) and the likely practicalities involved, making it highly unlikely that site specific estimates of such attributes could be obtained for the purposes of an assessment.

The population models produced for the examples in this report also derive from existing demographic data, and the reliance on existing data for this purpose is likely to be necessary in any application of this modelling approach (given the intensive and long-term nature of the studies required to generate such data). In the examples presented here, the demographic data derive from the long-term studies on the Isle of May, and there is at least some basis for assuming that these data are representative of the areas around the development site, with the Isle of May and Pentland Firth occurring within the same broad regional cluster of the UK for population trends of both shag and guillemot, if not for breeding success (Cook & Robinson 2010). Such detailed data on seabird demographics are available from relatively few sites in the UK, and in many instances there may be considerable challenges in producing reliable estimates of 'acceptable' levels of additional mortality. However, where available, population trend data can be used to assess the reliability of the population models from which 'acceptable' levels of additional mortality are estimated, and potentially allow modification of the models to reflect these trends. The availability of demographic data

also varies between species, and although high quality data are available for many of the species likely to be considered at risk from tidal developments (Macleay *et al.* 2007), for others data may be sparse, so limiting the potential for population modelling (e.g. red-throated diver *Gavia stellata*).

Another important assumption concerns the estimation of the source population size. In the examples used here this is achieved by applying generic foraging range data to identify the colonies that may contribute birds to the site, and subsequently using the available count data from these colonies to provide estimates of abundance. Thus, estimation of the 'source' colonies is coarse-grained, whilst in some situations the actual colony count data may be a decade or more old and potentially unrepresentative of current abundance. At certain sites these may remain limitations of the approach but at others there may be scope for refinement through basic additional data collection (e.g. systematic recording of flightlines of departing birds that have been foraging on the development site) or by considering a range of potential source colonies (e.g. from assuming that all birds are from the closest large colony to assuming birds are derived from all colonies within foraging range). Given the sensitivity of the T to variation in the source population size, the latter of these options will sometimes result in a wide range of possible values of T .

The sensitivity analyses undertaken in this report also indicate the importance of precision in estimating the number of birds that use the development site (Figure 5), and increasing survey effort is likely to be a valuable (and practical) means of reducing uncertainty in model outputs, where the available resources permit this. Further options to improve the reliability of model outputs may include examining how temporal variation in the use of the development site and diving activity by the species of interest relates to tidal flows and associated variation in turbine rotation speeds.

Therefore, it is apparent that there are ways to improve the confidence in the predictions from the modelling approach that is presented here, primarily through undertaking targeted data collection on the development site. However, there are also real constraints in the extent to which such additional data collection is feasible, particularly within the scope of the work that is likely to be undertaken for ornithological assessments, and even with the replacement of generic data with those derived from the site, modelling outputs will remain relatively broad based.

5.2 A consideration of the nature of the model outputs

The modelling approach described in this report produces outputs that are expressed in terms of the collision rate required to achieve a threshold level of additional mortality, as opposed to producing an actual figure for the number of collisions that are predicted to occur within a given time period, as is typical of more standard approaches to collision risk modelling (Band *et al.* 2007). The subsequent interpretation of outputs from the exposure time model is based upon a subjective assessment of whether the required collision rate is likely to occur or not.

In terms of providing a basis for making an assessment of impacts, this may at first appear less satisfactory than producing an actual figure for the predicted collision rate. However, in this respect it is worth considering the limited knowledge base and poor understanding of the underwater movements of diving birds and their behavioural responses to underwater devices, including their ability to evade the rotating turbines. Therefore, as detailed above, in the vast majority of situations it is likely to be possible to make broad based assessments of the potential impacts from collisions only, around which there is likely to remain high levels of uncertainty. In these circumstances there may be disadvantages to producing an estimate of the actual collision rate due to the potential for such estimates to be regarded in a definitive way. Furthermore, such estimates of the collision rate are likely to depend upon the use of

avoidance rates, for which there is currently little (or no) empirical basis but which would add substantially to the uncertainty in the model outputs (Chamberlain *et al.* 2006).

It is likely that there will be further potential to develop collision modelling frameworks for tidal turbines in the near future and to resolve some of the key factors currently limiting the ability to reliably predict actual collision rates (e.g. in terms of avoidance rates). This may arise through investigating and developing alternative modelling approaches (e.g. encounter rate modelling – Wilson *et al.* 2007), or through gaining a greater understanding of the behavioural responses of diving birds to such devices following their deployment in the marine environment and the initiation of associated studies to investigate their effects on diving birds. In this event there is scope to adapt the exposure time approach detailed in this report to provide the measure of bird activity within the development site from which subsequent collision rates can be estimated.

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Policy and Advice Directorate, Great Glen House,
Leachkin Road, Inverness IV3 8NW
T: 01463 725000

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