Population Viability Analysis and Comparison of Two Monitoring Strategies for Bottlenose Dolphins (*Tursiops truncatus*) in the Shannon Estuary, Ireland, to Inform Management

Miguel Blázquez,[1,2] Isabel Baker,[2] Joanne M. O'Brien,[1,2] and Simon D. Berrow[1,2]

[1] *Irish Whale and Dolphin Group, Merchants Quay, Kilrush, County Clare, Ireland*
E-mail: miguelblazvas5@gmail.com
[2] *Marine and Fresh Water Research Centre, Galway-Mayo Institute of Technology, Dublin Road, Galway, Ireland*

Abstract

Photo-identification has been used to monitor the resident bottlenose dolphin (*Tursiops truncatus*) population in the Lower River Shannon Special Area of Conservation in Ireland as part of European Union obligations since 1997. A discovery curve using data collected by the Irish Whale and Dolphin Group (IWDG) between 2011 and 2015 suggested all animals in the population were captured during this period, providing an abundance of 145 extant individuals. The IWDG photo-identification catalogue was compared with an independently derived photo-identification catalogue compiled by the National Parks and Wildlife Service during 2015 to identify whether any individuals were not captured by the IWDG during that year; there were none. Mark-recapture analysis of both photo-identification catalogues indicated that the higher survey effort provided a higher and more precise abundance estimate, and the estimate of 145 individuals in the population was supported. Population Viability Analysis using this most up-to-date abundance estimate and new data on life-history characteristics were used to explore population status under a range of potential scenarios. A baseline model showed slowly declining dynamics with a growth rate (r) of -0.007. An increase in adult mortality from 3.8 to 7.0% (e.g., due to bycatch) led to a steep decline in population size. Similar results were obtained when the effects of an infrequent catastrophic event (e.g., a large oil spill) were simulated. Sensitivity analysis revealed the importance of female reproductive capacity for population viability. These results will help local managers to understand the implications of events that may affect this small, genetically discrete resident population. The current monitoring strategy may be insufficient to obtain a thorough understanding of population dynamics; hence, higher survey effort is recommended to gain a better knowledge of key demographic parameters which may affect population viability, especially those limiting female reproductive capacity.

Key Words: bottlenose dolphin, *Tursiops truncatus*, photo-identification, mark-recapture, abundance, population viability, simulation, sensitivity analysis

Introduction

The common bottlenose dolphin (*Tursiops truncatus*; Montagu, 1821) is a cosmopolitan species found in all oceans at temperate and tropical latitudes, typically using coastal shallow ecosystems such as bays, estuaries, and rivers (Leatherwood & Reeves, 1983), although there are also offshore pelagic ecotypes (Wells & Scott, 1999). Coastal ecotypes tend to be restricted in their range and movement and usually establish long-term and multi-generational home ranges (Leatherwood & Reeves, 1983; Wells & Scott, 1990; Hammond et al., 2012), which facilitates their study and monitoring through photo-identification and mark-recapture techniques (Berrow et al., 2012).

In the eastern North Atlantic Ocean, bottlenose dolphins have a patchy distribution, including several coastal populations present along the western seaboard of Europe (Dos Santos & Lacerda, 1987; Berrow et al., 1996; Wilson et al., 1999; Evans et al., 2003; Kiszka et al., 2004; Methion & Díaz López, 2018). One of these locations, the Shannon Estuary, located on the west coast of Ireland, was designated as a Special Area of Conservation (SAC) in 2000 to maintain the bottlenose dolphin population at a Favourable Conservation Status (Epstein et al., 2016). A number of studies have been carried out in the area since the early 1990s (Berrow et al., 1996, 2012; Ingram, 2000; Rogan et al., 2000, 2003; Ingram & Rogan, 2002, 2003; Englund et al., 2007, 2008; Berrow, 2009; Foley et al., 2010; Mirimin et al., 2011), which established the singularity of this bottlenose dolphin

population, including residency, site preference, small population size, home range, and genetic isolation. These early studies supported the designation of the Shannon Estuary as a SAC for bottlenose dolphins.

Triennial mark-recapture studies, funded by the National Parks and Wildlife Service (NPWS) of Ireland, indicate the population has remained relatively stable over the last two decades (Ingram, 2000; Ingram & Rogan, 2003; Englund et al., 2007, 2008; Berrow et al., 2012; Rogan et al., 2015, 2018). Englund et al. (2008) suggested that this triennial monitoring strategy was not powerful enough to detect significant changes in the population in a reasonable timeframe. In parallel, the Irish Whale and Dolphin Group (IWDG) has been conducting monitoring and research on this bottlenose dolphin population since 1993 (Berrow et al., 1996), with an increase in annual research effort since 2011 (Berrow et al., 2016; Baker et al., 2018a, 2018b). As a result, a discovery curve showed that, following intensive survey effort, no new individuals were added to the IWDG photo-identification catalogue in 2015 (Baker et al., 2018a), which suggested all individuals in the population were identified. Data collected during the 2015 NPWS triennial survey provided an opportunity to test this assertion. Therefore, photo-identification data collected in the Shannon Estuary in 2015, both by the NPWS and the IWDG, were used to investigate the abundance of bottlenose dolphins in the Shannon Estuary and to compare both triennial and annual monitoring strategies.

Recent studies have expanded our knowledge of critical aspects of this population such as social structure and demographic parameters (Baker et al., 2018a, 2018b). This new information provided an opportunity to explore population status and evaluate current management strategies. Population Viability Analysis, or PVA, is a set of methods of quantitative analysis used for evaluating threats to wildlife populations, their risk of extinction, and their chances for decline or recovery, based on population- and species-specific data (Akçakaya & Sjögren-Gulve, 2000; Lacy, 2019). PVA is usually performed throughout simulation models to project population trajectories with the aim to guide decision-making and management of a species or population, and thus, it has become an essential methodology in conservation science in the last decades (Lacy, 2019).

Due to their proximity to human development, small population size, and home range, coastal populations of bottlenose dolphins are typically exposed to numerous anthropogenic threats. These include the toxic effects of xenobiotic chemicals, reduced prey availability due to habitat degradation and overfishing, direct or indirect disturbance

and harassment, marine construction or demolition, and other forms of environmental degradation such as underwater noise (Hammond et al., 2012). Many of these threats could potentially affect the bottlenose dolphin resident population in the Shannon Estuary (Figure 1). Due to its depth and shelter, the estuary provides ideal shipping access to the largest vessels entering Irish waters (180,000 to 200,000 deadweight tonnage) while servicing six main terminals and handling up to 1,000 ships per annum carrying 10 to 12 million tons of cargo (Rogan et al., 2000; O'Brien et al., 2016; Limerick City and County Council, 2019). Thus, the estuary is an important shipping route, especially towards the ports at Foynes and Limerick (National Parks and Wildlife Service [NPWS], 2012). In addition, the Shannon Estuary is a major centre of industry with an alumina smelting plant at Aughinish and two power stations located at Money Point and Tarbert in the middle-estuary. Also, The River Shannon catchment includes large areas of farmland and several tributary rivers providing additional sources of contamination (Berrow et al., 2002; Jepson & Law, 2016). Given this situation, PVA can serve as a powerful tool to assess the potential effects that these threats could have on the bottlenose dolphins inhabiting the Shannon Estuary.

PVA can be used to forecast how the population is likely to change through time, given some demographic traits, or to evaluate potential threats, but it can also be applied to provide guidance to which management actions can optimize conservation outcomes through sensitivity analysis (Manlik et al., 2018). Sensitivity analysis is a collection of analytical tools used to assess how changes in vital rates of a demographic model or simulation affect population growth or risk of extinction (Mills & Lindberg, 2002; Manlik et al., 2018) and has been utilized to guide conservation actions for a wide range of species (Pulliam et al., 1992; Gong et al., 2012; Manlik et al., 2016; Lacy et al., 2017; Fantle-Lepczyk et al., 2018a, 2018b). In our case study, sensitivity analysis was applied to determine what aspects of the biology of these bottlenose dolphins are the most important for their population viability and which of these need to be studied in more detail to inform future research and improve current management strategies.

The objectives of this study were (1) to assess current monitoring strategies of the resident bottlenose dolphin population in the Lower River Shannon SAC, (2) to conduct PVA to assess the viability of this bottlenose dolphin population, and (3) to evaluate the potential effect of different threats on the viability of the population and to determine priorities for research and management to maintain a Favourable Conservation Status.
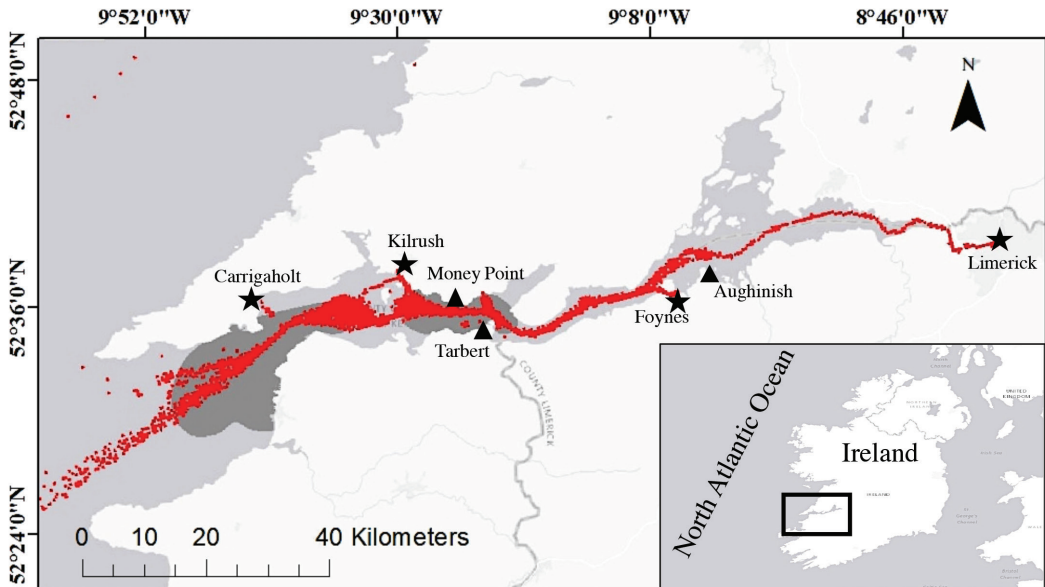
Figure 1. Map showing the Shannon Estuary in Ireland, including main harbours and ports (stars), industrial plants and power stations (triangles), bottlenose dolphin (*Tursiops truncatus*) critical areas (dark grey zones), and main shipping channels (dots trail) (Marine Institute, www.marine.ie)

Methods

Study Area

The Shannon Estuary, which is the longest waterway in Ireland (*ca* 80 km), presents a convoluted geometry that is narrowly elongated on a southwest-northeast axis with a complex bathymetry (O'Brien et al., 2016). Due to the intense influence of tides, the estuary has been modified historically by a hydroelectric scheme (Rogan et al., 2000). Bottlenose dolphins occur at two main areas (Figure 1) with large slope and depth where they have been seen foraging (Ingram & Rogan, 2002; Englund et al., 2007, 2008).

Comparison of Photo-Identification Catalogues

To compare both bottlenose dolphin monitoring programmes currently running in the Shannon Estuary, we used photo-identification data collected by the NPWS and the IWDG in 2015 since this year was combined with simultaneous research effort by the NPWS (Rogan et al., 2015) and the IWDG (Baker et al., 2018a).

Baker et al. (2018a), using data collected between 2011 and 2015, showed that no previously unidentified bottlenose dolphins were recorded in 2015, suggesting all individuals in the population were photographed with a discovery curve that reached a clear plateau. We analysed both dolphin fin identification catalogues to determine if the NPWS catalogue, which was smaller than the IWDG catalogue, included individuals not photographed by IWDG researchers in 2015. Each individual identified in the NPWS catalogue was compared and matched with each individual identified by the IWDG. Individual bottlenose dolphins can be recognised through scars and wounds they acquire, particularly on their dorsal fin, allowing each individual to be identified for extended periods of time, ranging from months to years (Würsig & Würsig, 1977; Leatherwood & Reeves, 1983). Each identified dolphin was compared considering as many features as possible, including fin shape; size and position of nicks and notches; colour and shape of scars; and other skin lesions, spots, stains, or other marks.

Mark-Recapture Analysis

In addition to cross-matching both of these photo-identification catalogues, we used both datasets to derive new mark-recapture abundance estimates. Individual bottlenose dolphins were classified according to the extent of their natural marks following Ingram (2000):

- *Severity Grade 1* – Marks consisting of significant fin damage or deep scarring that can be considered permanent

- *Severity Grade 2* – Marks consisting of deep tooth rakes and lesions and minor cuts

- *Severity Grade 3* – Marks consisting of very superficial lesions or complete absence of them

Multiple sample mark-recapture abundance estimates of closed populations depend on the following four assumptions:

1. Population is closed.

2. Animals do not lose their marks during the experiment.

3. All marks are correctly noted and recorded at each trapping occasion.

4. Each individual has a constant and equal probability of capture on each trapping occasion.

Photo-identification data collected by the NPWS and the IWDG in 2015 were used to create two separate sighting histories for each dataset. These sighting histories only included bottlenose dolphins classified as Severity Grade 1. This was applied to meet Assumptions 2 and 3 (Wilson et al., 1999). Additionally, calves were excluded as the majority of them did not present permanent marks and they tend to be strongly associated with their mothers (Wilson et al., 1999), which violates Assumption 4. It has been discussed that Assumption 4 does not tend to occur in natural systems (Otis et al., 1978). Due to this fact, the mark-recapture dedicated software, *CAPTURE* (Rexsted & Burnham, 1991), includes more flexible mark-recapture models which allow inconstance and unequal probability of capture of individuals (White et al., 1982). Thus, tests to determine the most appropriate model were applied. Assumption 1 (i.e., population closure; no recruitment or losses occur) is generally defined to mean that there are no unknown changes to the initial population throughout the whole study (White et al., 1982). This is a strong assumption, but it can be met, at least approximately, if the study is correctly designed. Therefore, *CAPTURE*'s test to assess population closure was also applied for each sighting history.

Additionally, only one estimate was derived for each dataset since the IWDG photo-identification catalogue contained images from both dorsal fin sides (right and left) of all individuals identified by the NPWS and the IWDG in 2015.

Both sighting histories were input into *CAPTURE*, Version 090210.1141, which provides an estimate of the number of permanently marked individuals which are a subset of the whole population. To estimate the total population size, this formula was applied:

$$N_{total} = \frac{N_{hat}}{\theta}$$

where N_{total} is the estimated total population size, N_{hat} is *CAPTURE*'s estimate of the number of animals with long-lasting marks, and θ is the estimated proportion of animals with long-lasting marks in the population.

Variance of N_{total} was estimated using the delta method as recommended by Wilson et al. (1999):

$$var(N_{total}) = N_{total}^2(\frac{var(N_{hat})}{N_{hat}^2} + \frac{(1-\theta)}{n\theta})$$

where n is the total number of animals from which θ was calculated; $Var = SE^2$.

PVA Software
PVAs were conducted using *Vortex*, Version 10.3.6.0 (Lacy, 1993), which generates Monte Carlo simulations of demographic and environmental processes. *Vortex* is able to run individual-based simulations, following each individual in the simulated population from birth to death with all events happening according to pre-defined probabilities. Events simulated by *Vortex* correspond with those occurring during the annual cycle of a sexual diploid organism such as mating, birth and death, increment of age by each year, migration among populations (metapopulation), removal or supplementation of individuals, and so on. *Vortex* is designed to model long-lived species with low fecundity such as the bottlenose dolphin (Leatherwood & Reeves, 1990).

Demographic Data and Baseline Model Parameterization
Demographic models were parameterized using recent data for the bottlenose dolphins in (Baker et al., 2018a), or as close as possible to, the Shannon Estuary to create the most accurate PVA models for this population to date (Table 1). A 50-y projection with 1,000 iterations (replicates) was used in the model.

The baseline model input (Table 1) was defined as the following: inbreeding depression was set at 0 since Mirimin et al. (2011) reported no evidence of inbreeding in the bottlenose dolphin population in the Shannon Estuary. "Polygamous mating" and "75% of males in the breeding pool" were chosen following Thompson et al. (2000), who used numbers from Wells & Scott (1990) for the same species off northeast Scotland. Age at first reproduction was based on Sergeant et al. (1973). Lifespan was set at 50 y on the basis of Read et al. (1993), and individuals were assumed to reproduce throughout

Table 1. *Vortex* input values (and source) for baseline and sensitivity analysis models for the bottlenose dolphin (*Tursiops truncatus*) population in the Shannon Estuary. EV (SD) = standard deviation due to environmental variation.

Parameter	Baseline values	Sensitivity values					Data from
		-50%	-25%	0%	+25%	+50%	
Inbreeding depression	No			Yes			Mirimim et al., 2011
Lethal equivalents		3.15	4.71	6.29	7.86	9.44	O'Grady et al., 2006
% due to recessive lethals				50			
Reproductive system							
Reproductive system	Polygamous			--			Thompson et al., 2000
Age of 1st offspring females	10			--			Sergeant et al., 1973
Age of 1st offspring males	11			--			Sergeant et al., 1973
Lifespan	50	20	30	40	50	60	Read et al., 1993
Max. age of reproduction	50	20	30	40	50	60	
Max. broods/year	1			--			Baker et al., 2018a
Max. progeny/brood	1			--			Baker et al., 2018a
Sex ratio at birth in % males	50			--			Leatherwood & Reeves, 1983
Density dependent reproduction	No			--			Thompson et al., 2000
Dependency of offspring (years)	3			--			Baker et al., 2018a
Reproductive rates							
% adult females breeding	57.2	28.6	42.9	57.2	71.5	85.8	Baker et al., 2018a
EV (SD)	2.57			--			
Distribution of broods/year	0-74%; 1-26%	1-13	20	26	33	39	Baker et al., 2018a
1 offspring/female/brood	100%			--			
Mortality rates as % (same for males and females)							
Mortality 0-1 year	11	5.5	8.25	11	13.75	16.5	Baker et al., 2018a
EV (SD)	4			-			
Mortality 1-50 years	3.8	1.9	2.85	3.8	4.75	5.7	Baker et al., 2018a
EV (SD)	1.5			--			
Mate monopolization				--			
% males in breeding pool	75%			--			Thompson et al., 2000
Initial population size							
Stable age distribution	Yes			--			
Initial population size	145	73	109	145	181	218	Baker et al., 2018a
Carrying capacity	290	146	218	290	362	436	Thompson et al., 2000
EV (SD)	0						

their whole life. Number of broods per year, number of progenies per brood, and the sex ratio at birth was set according to empirical observation (Baker et al., 2018a) and general knowledge of the species (Leatherwood & Reeves, 1983). With respect to reproductive rates, the proportion (%) of females breeding was calculated as the mean ratio between the number of reproductive females identified each year divided by the total number of identified adult females within the interval 2011 to 2015 from Baker et al. (2018a). The % of reproductive females having one calf per year was 26%, provided by Baker et al. (2018a) for this population fecundity (surviving calves/reproductive female/year). Duration of offspring dependency was based on inter-birth intervals (Baker et al., 2018a). Mortality rates for both first-year calves and adults (adults encompass juveniles) were also based on Baker et al. (2018a). In addition, carrying capacity was set as double the initial population size (Thompson et al., 2000; Englund et al., 2008). Since *Vortex* requires us to include the SD of carrying capacity due to Environmental Variation (EV) as input

value, it was set at 0 as there are no data for the Shannon Estuary; however, this is not an issue as the carrying capacity will not be reached. Moreover, although age proportions are known for this population, sex distribution is less known since less than a half of all identified individuals were sexed (Baker et al., 2018a). Hence, a stable age distribution was applied.

Sensitivity Analysis
The baseline model was used as a reference from which a sensitivity analysis was performed to understand the effect of key parameters on model outcomes (Akçakaya & Sjögren-Gulve, 2000; Mills & Lindberg, 2002; Manlik et al., 2018). Each key parameter (Tables 1 & 5) was varied ±25% and ±50% holding all other values of the baseline model each time following Fantle-Lepczyk et al. (2018b). We examined lifespan, inbreeding, initial population size, % of females breeding, distribution of broods per year, mortality of first-year calves, mortality of adult females, and mortality of adult males. The standard sensitivity index (S_x) (Morris & Doak, 2002; Fantle-Lepczyk et al., 2018a, 2018b) was used for this analysis and was calculated as

$$S_x = \frac{X_{new} - X_{baseline}}{P_{new} - P_{baseline}}$$

where *X* is a *Vortex* output value (stochastic r, stochastic λ, or N-extant) and *P* is the parameter of interest (Fantle-Lepczyk et al., 2018a, 2018b). The standard sensitivity index measures the degree of difference between modified and baseline values. The further the index is from 0, the higher the sensitivity is.

It is important to note that two parameters assessed in the sensitivity analysis were not based on the baseline model: (1) inbreeding and (2) lifespan (Table 1). Inbreeding was set at 0 in the baseline model since there was no evidence of inbreeding in the bottlenose dolphin population in the Shannon Estuary according to Mirimin et al. (2011). To test the effect of inbreeding, the base value was centred at 6.29 lethal equivalents which is the mean combined effect of inbreeding on fecundity and first-year survival in a survey of impacts on wild species (O'Grady et al., 2006). Regarding lifespan, the base value was centred at 40 y instead of 50. This was done to include within the interval ±50% ages that bottlenose dolphins are known to reach. Bottlenose dolphins have been reported to live less than 30 y in captivity (Jaakkola & Willis, 2019), whereas they can live longer in the wild, reaching at least 31 y in the Moray Firth in northeast Scotland (B. Cheney, pers. comm., 5 April 2019) and 64 y in Sarasota Bay, Florida

(Irvine, 2019). These changes with respect to the baseline model should not affect the sensitivity analysis output as it measures the effect of modifying parameters within the same proportions (±25% and ±50%).

Hypothetical Scenarios to Explore the Potential Impacts of Threats
Additional models were run to explore the effects of a series of potential scenarios on population dynamics and to compare them with the baseline model. The main objective of this was to compare the long-term effects of events that, hypothetically, may take place in the Shannon Estuary. We simulated a reduction in lifespan; an increase in annual adult mortality due to bycatch, for example; an increase in first-year calf mortality; a catastrophic oil spill decreasing both reproduction and survival rates; and a small increase in female survival due to an increase in research effort (Table 2).

Reduced Lifespan—Bottlenose dolphin lifespan in the wild is generally accepted to be around 50 y (Sergeant et al., 1973), with some known individuals reported to exceed 60 y (Irvine, 2019). However, these life expectancies come from populations inhabiting low latitudes and have not been tested in higher latitudes such as Ireland. Currently, the only approach to estimate the lifespan of bottlenose dolphins in the Shannon Estuary is that some individuals identified in the early 1990s (approx. 30 y ago) for the first time are still alive and frequently sighted in the estuary nowadays. The same lifespan has been reported for an adult female from the Moray Firth (B. Cheney, pers. comm., 5 April 2019), an area which is very similar to the Shannon Estuary in terms of latitude and population size (Arso-Civil et al., 2019; Cheney et al., 2019). Thus, for this scenario, we modelled the effect of population viability on a maximum lifespan of 30 y.

Increased Adult Mortality Due to Bycatch—Bottlenose dolphins may be caught in gillnets and occasionally in pelagic trawls (Berrow & Rogan, 1998; Morizur et al., 1999). The Shannon Estuary is not characterised by intense fishing activity; however, it is higher in nearby areas off the southwest coast of Ireland (Gerritsen & Lordan, 2014). Moreover, it is known that Shannon dolphins can occur in other coastal areas out of the Shannon Estuary (Mirimin et al., 2011; Levesque et al., 2016) where they are likely to interact with these fisheries.

Although mortality causes in the Shannon Estuary remain unclear, and assuming that current mortality estimates are not caused by bycatch-derived deaths, this model explored the effect of a small increase in current adult mortality estimate (3.8%) up to 7%, which is equivalent to two additional individuals dead per year due to incidental

Table 2. *Vortex* input values for hypothetical scenario models including values that were modified regarding the baseline model. Note that large oil spill model's values are input by multiplying survival rates (100 - mortality rate) by 0.5 (50%) and reproductive rates by 0.2 (20%).

Model parameter	Baseline	Reduced lifespan	Increased adult mortality (bycatch)	Increased 1st-year calf mortality	Catastrophic oil spill	Reduced female mortality
Lifespan	50	30	--	--	--	--
Adult mortality	3.8	--	7	--	51.9	--
1st-year calf mortality	11	--	--	20	55.5	--
Female mortality	3.8	--	--	--	51.9	2.5
% breeding	60	--	--	--	12	--
Broods distribution	1-26%, 0-74%	--	--	--	1-5.2%, 0-94.8%	--

bycatch or entanglement from interaction with fisheries. Similar bycatch rates have been reported for the larger bottlenose dolphin population interacting with a trawl fishery in Pilbara (Western Australia) where between 1.13 and 2.17% of the individuals were killed by bycatch per year (Fletcher & Santoro, 2012; Wakefield et al., 2014; Allen et al., 2017).

Increased First-Year Calf Mortality—Current mortality rate for first-year calves (11%) is a low value compared with those from other populations (Wells & Scott, 1990; Manlik et al., 2016; Arso-Civil et al., 2019). Baker et al. (2018a) warned that the sample size used to calculate this value was small, and mortality rates are likely higher. As in the adults' case, calf mortality causes are largely unknown in the Shannon Estuary. Therefore, for this scenario, we explored the effect on population viability of a significant increase in first-year calf mortality substituting the current mortality value (11%) by 20%, which has been observed in the bottlenose dolphin population in Sarasota Bay, Florida (Wells & Scott, 1990).

Catastrophic Oil Spill—Due to its length, depth, and protection, the Shannon Estuary is one of the busiest waterways in Ireland; it is used by some of the largest vessels entering Irish waters (Rogan et al., 2000; O'Brien et al., 2016; Limerick City and County Council, 2019) and is expanding (Shannon Foynes Port Company [SFPC], 2013). Therefore, there is a possibility of incidental oil spills or other toxic substances in the area, which may have devastating effects on the bottlenose dolphin population and its habitat. Since there have been no catastrophic oil spills in the Shannon Estuary to date, only small local spills, this scenario simulated that a large oil spill would occur in the Shannon Estuary with a 5% frequency, equivalent to one catastrophe every 20 y over a 50-y projection, with similar effects on reproduction and mortality to those observed in offshore bottlenose dolphin populations in the Gulf of Mexico after the *Deepwater Horizon* oil spill in 2010 (Lane et al., 2015; McDonald et al., 2017).

Reduced Adult Female Mortality—Adult mortality value used in the baseline model (3.8%) was calculated by Baker et al. (2018a), assuming that all individual bottlenose dolphins that were not sighted regularly in the Shannon Estuary for several years were actually dead. This means that 3.8% is a maximum mortality rate since some of these dolphins considered dead may actually have emigrated from the population (Ryan et al., 2010; Mirimin et al., 2011; Levesque et al., 2016; Nykänen et al., 2018) and may be resighted in the Shannon Estuary in the future. Hence, adult mortality, and particularly adult female mortality, may be actually lower than 3.8%. This value is not sex-specific; and while just a half (49%) of known individuals are sexed, most of them are females (82%). Being the only adult mortality estimate to date for this population, we assumed that a revised and more precise estimation of adult mortality would provide a lower value for female mortality. This last model explored the effect of a slight reduction in adult female mortality down to 2.5%, which is equivalent to avoiding the death of just one individual female per year according to current knowledge of population age structure (Baker et al., 2018a).

The extent of the impacts of these hypothetical scenarios, while speculative in some cases, do serve to show the potentially significant effects of anthropogenic factors.

Results

Analysis of Both Photo-Identification Catalogues
The NPWS photo-identification catalogue contained 100 uniquely marked individual bottlenose dolphins identified during 12 complete surveys carried out between June and October 2015. The IWDG photo-identification catalogue contained 136 individuals identified during 77 complete surveys from April to September in the same year. Both catalogues were cross-matched to verify whether there were any individuals identified by the NPWS but not by the IWDG, in spite of the lower survey effort by the NPWS. No dolphin unidentified by the IWDG was identified during the NPWS surveys. Moreover, a number of false positives (two individuals were actually the same one) were found in the NPWS catalogue. It was determined that the NPWS catalogue actually contained 85 identified dolphins, not 100. No errors were found in the IWDG catalogue. False positive matches may lead to an overestimation of mark-recapture estimates (Gunnlaugsson & Sigurjónsson, 1990; Yoshizaki et al., 2009; Urian et al., 2015), so a new mark-recapture estimate was derived using the reviewed NPWS catalogue as well as the IWDG catalogue.

Mark-Recapture Analysis
Mark-recapture abundance estimates were calculated using identified bottlenose dolphins from both catalogues which presented long-lasting marks, excluding calves. Both sighting histories were found to be closed according to the *CAPTURE* closure test in which the null hypothesis is closure + model Mh (NPWS, $p = 0.705$; IWDG, $p = 0.190$). The most appropriate model for the NPWS dataset was Mt, whereas the most appropriate one for the IWDG dataset was Mtb (for details, see Otis et al., 1978; White et al., 1982). Therefore, these two models were applied for estimating Severity Grade 1 dolphin abundance from their respective sighting history. These estimates of marked individuals (N_{hat}) were inflated according to the estimated proportion of marked individuals in the population or θ (Table 3).

The revised NPWS photo-identification catalogue resulted in a lower abundance estimate (93 ± 8.81; Table 4) compared with that calculated by Rogan et al. (2015) (114 ± 13.5). In addition, this revised estimate was considerably lower than the estimate from the IWDG sighting history (Table 4) and the minimum abundance estimate calculated by Baker et al. (2018a) in 2015 of 145 extant individuals. This last value was contained within the 95% CI for the mark-recapture estimate of the IWDG photo-identification catalogue in 2015 (Table 4). Moreover, the IWDG estimate of 136 is almost the same number of individuals (135) contained in that photo-identification catalogue from 2015 (Baker et al., 2018a).

Sensitivity Analysis
Lower sensitivity was yielded by adult male mortality, inbreeding, and initial population size on both population growth rate (r) and final population size (N-Extant) (Table 5). Mortality of first-year calves had a slightly higher effect on models' sensitivity than adult male mortality, inbreeding, and initial population size. On the other hand, models had high sensitivity to mortality of adult females, lifespan, and reproductive rates (% of

Table 3. Proportion of bottlenose dolphin identifications with long-lasting marks for the revised National Parks and Wildlife Service (NPWS) and the Irish Whale Dolphin Group (IWDG) photo-identification catalogues in 2015. θ is the estimated proportion of dolphins with long-lasting marks.

| Catalogue | Long-lasting marks | | Total | θ |
	With	Without		
NPWS	149	107	256	0.58
IWDG	440	365	805	0.55

Table 4. Mark-recapture abundance estimates of bottlenose dolphins in the Shannon Estuary in 2015 for the revised NPWS and the IWDG photo-identification catalogues. N_{hat} = estimated number of dolphins with long-lasting marks; θ is the estimated proportion of dolphins with long-lasting marks; and SE = standard error.

Catalogue	N_{hat}	θ	Abundance estimate	SE	Coefficient of variation	Abundance 95% CI
NPWS	54	0.58	93	8.81	0.09	83-103
IWDG	75	0.55	136	18.03	0.13	125-202

females breeding and distribution of broods per year). In general terms, mortality of adult females was the most influential parameter on either population growth rate or final population size at all proportions of change (Table 5).

Hypothetical Scenarios to Explore the Potential Impacts of Threats
With new knowledge on the key parameters which had the greatest effect on population viability according to the sensitivity analysis, a series of additional models were constructed to simulate the fate of the population under different management scenarios.

The baseline model yielded by *Vortex* indicated that, with current values, the bottlenose dolphin population in the Shannon Estuary would be slightly declining (r = -0.007; Figure 2 & Table 6). A reduction of baseline lifespan (50 y) down to 30 y would lead to a population decline (r = -0.0334; Figure 2 & Table 6).

An increase in annual adult mortality from 3.8 to 7% (e.g., due to fisheries bycatch) would lead to a steeper decline (r = -0.447; Figure 2 & Table 6). This means that the additional mortality of just two dead individuals each year would have catastrophic effects on the population in the long term. Further, an increase in first-year calf mortality rates from 11 to 20% would have a weaker effect than increased adult mortality and reduced lifespan (r = -0.0113; Figure 2 & Table 6).

Since the Shannon Estuary is an important shipping channel used by a high and increasing number of large vessels, there is the possibility of damaging oil spills in the area. A large oil spill happening with a frequency of 5% (once every 20 y) would reduce survival rates by 50% and reproduction rates by 20%, leading to a very steep decline (r = -0.0451; Figure 2 & Table 6). On the other hand, if an increase in research effort provided a lower but more precise female adult mortality estimate of 2.5%, this new estimate would lead the population to a moderate positive growth (r = 0.012; Figure 2 & Table 6).

Discussion

Comparison of Both Photo-Identification Catalogues
Baker et al. (2018a) suggested the Shannon Estuary bottlenose dolphin population was comprised of 145 extant individuals. This was based on a discovery curve created over a 4-y period and the absence of new individuals recorded in 2015 despite a high survey effort. Rogan et al. (2015) identified 100 uniquely marked individual dolphins; however, a number of false positive matches were found, resulting in a revised

number of 85 identified individuals. These false identifications created the so-called "ghost capture histories" (Yoshizaki et al., 2009), which led to positively biased mark-recapture estimates (Gunnlaugsson & Sigurjónsson, 1990; Yoshizaki et al., 2009; Urian et al., 2015). It has been well documented that rates of matching errors increase as photographic quality decreases (Stevick et al., 2001; Friday et al., 2008; Frasier et al., 2009; Barlow et al., 2011). Thus, it is very important to define a threshold for photographic quality in mark-recapture studies since models assume that every individual is identifiable and correctly identified (Urian et al., 2015). Rogan et al. (2015) did establish a threshold for photographic quality; however, the quality of some photographs selected as part of their photo-identification catalogue were low enough to lead to matching errors. Low-quality photographs can be a result of several factors such as inexperienced photographers, unsuitable weather or high sea state, inappropriate light conditions, or poor equipment quality. Additionally, low survey effort may also result in the use of low-quality photographs to boost the potential number of captures. The more dedicated the effort carried out during photo-identification surveys, the larger the amount of data collected and, thus, the higher probability of obtaining good quality photographs. During 2015, only 12 photo-identification surveys were carried out by the NPWS in the Shannon Estuary (Rogan et al., 2015), whereas the IWDG carried out a total of 91 surveys, identifying dolphins during 77 of them during the same period. This demonstrates that a high survey effort is desirable to obtain sufficient high-quality photo-identification data to derive robust abundance estimates.

Mark-Recapture Analysis
Rogan et al.'s (2015) mark-recapture estimate for bottlenose dolphins in the Shannon Estuary in 2015 was 114 ± 13.5, CV = 0.12 (95% CI = 90 to 143). This estimate was positively biased due to false positive matchings and, thus, a new estimate of 93 ± 8.81, CV = 0.09 (95% CI = 83 to 103) was calculated. A mark-recapture analysis was carried out on the IWDG photo-identification catalogue during the same time period, and an estimate of 136 ± 18.03, CV = 0.13 (95% CI = 125 to 202) was calculated (Table 4). The IWDG estimate (136) was closer to the abundance estimate reported by Baker et al. (2018a) of 145 extant individuals in 2015, which is within the 95% CI (125 to 202). Interestingly, the abundance estimate of 136 is almost the same as the number of individuals in the IWDG catalogue for 2015. The difference between Baker et al.'s (2018a) estimate of 145 extant individuals and our mark-recapture

Table 5. *Vortex* output for sensitivity analysis models and standard sensitivity indices (S_x). r = stochastic growth rate; PE = probability of extinction (proportion of iterations in which the population went extinct); N-Extant = final population size; and SD = standard deviation.

| Models	Magnitude of variation	r	SD(r)	PE	N-Extant	SD (N-Extant)	S_x for r	S_x for N-Extant
Lifespan	-50%	-0.0877	0.1145	0.7530	4.03	1.91	0.0037	3.374
	-25%	-0.0333	0.0490	0.0010	29.63	11.06	0.0019	4.187
	0%	-0.0148	0.0341	0.0000	71.55	18.80	0	0
	+25%	-0.0069	0.0298	0.0000	105.61	24.76	0.0008	3.411
	+50%	-0.0028	0.0281	0.0000	128.75	25.96	0.0006	2.863
Inbreeding	-50%	-0.0071	0.0297	0.0000	104.30	23.53	0.0001	0.723
	-25%	-0.0073	0.0300	0.0000	103.24	23.58	0.0002	0.917
	0%	-0.0070	0.0300	0.0000	105.04	24.35	0	0
	+25%	-0.0073	0.0299	0.0000	103.03	22.63	-0.0002	-1.108
	+50%	-0.0073	0.0298	0.0000	103.59	23.54	-0.0000	-0.384
Initial population size	-50%	-0.0073	0.0420	0.0000	53.36	16.34	0.0000	0.13
	-25%	-0.0071	0.0342	0.0000	79.24	20.25	0.0000	0.14
	0%	-0.0070	0.0299	0.0000	104.99	23.87	0	0
	+25%	-0.0070	0.0270	0.0000	130.53	26.83	0.0000	0.143
	+50%	-0.0067	0.0247	0.0000	158.56	29.97	0.0000	0.155
% females breeding	-50%	-0.0317	0.0394	0.0010	31.24	9.24	0.0009	2.836
	-25%	-0.0168	0.0329	0.0000	64.67	16.10	0.0008	3.444
	0%	-0.0071	0.0294	0.0000	116.34	25.52	0	0
	+25%	0.0043	0.0273	0.0000	183.83	36.60	0.0006	4.499
	+50%	0.0120	0.0261	0.0000	255.36	33.84	0.0006	4.634
Distribution of broods	-50%	-0.0339	0.0406	0.0000	28.09	8.84	0.0020	5.858
	-25%	-0.0171	0.0329	0.0000	63.84	15.99	0.0017	6.735
	0%	-0.0071	0.0299	0.0000	104.25	23.05	0	0
	+25%	0.0030	0.0276	0.0000	172.20	35.09	0.0014	9.707
	+50%	0.0100	0.0264	0.0000	238.95	38.75	0.0013	10.364
Adult female mortality	-50%	0.0142	0.0225	0.0000	274.17	22.47	-0.0111	-88.76
	-25%	0.0037	0.0256	0.0000	177.69	33.63	-0.0112	-76.24
	0%	-0.0069	0.0298	0.0000	105.53	23.67	0	0
	+25%	-0.0176	0.0349	0.0000	62.74	17.17	-0.0112	-45.04
	+50%	-0.0286	0.0407	0.0020	36.58	11.65	-0.0114	-36.29
Adult male mortality	-50%	-0.0070	0.0263	0.0000	105.08	23.94	-0.0001	-0.258
	-25%	-0.0069	0.0285	0.0000	105.60	24.05	-0.0002	-1.063
	0%	-0.0071	0.0297	0.0000	104.59	23.93	0	0
	+25%	-0.0070	0.0310	0.0000	104.76	23.20	0.0001	0.179
	+50%	-0.0070	0.0320	0.0000	104.88	24.59	0.0001	0.152
1st-year calf mortality	-50%	-0.0046	0.0292	0.0000	118.34	26.36	0.0005	-2.495
	-25%	-0.0060	0.0297	0.0000	110.32	24.52	0.0004	-2.072
	0%	-0.0071	0.0299	0.0000	104.62	23.84	0	0
	+25%	-0.0085	0.0303	0.0000	97.41	22.41	-0.0005	-2.622
	+50%	-0.0096	0.0307	0.0000	92.16	21.50	-0.0005	-2.265
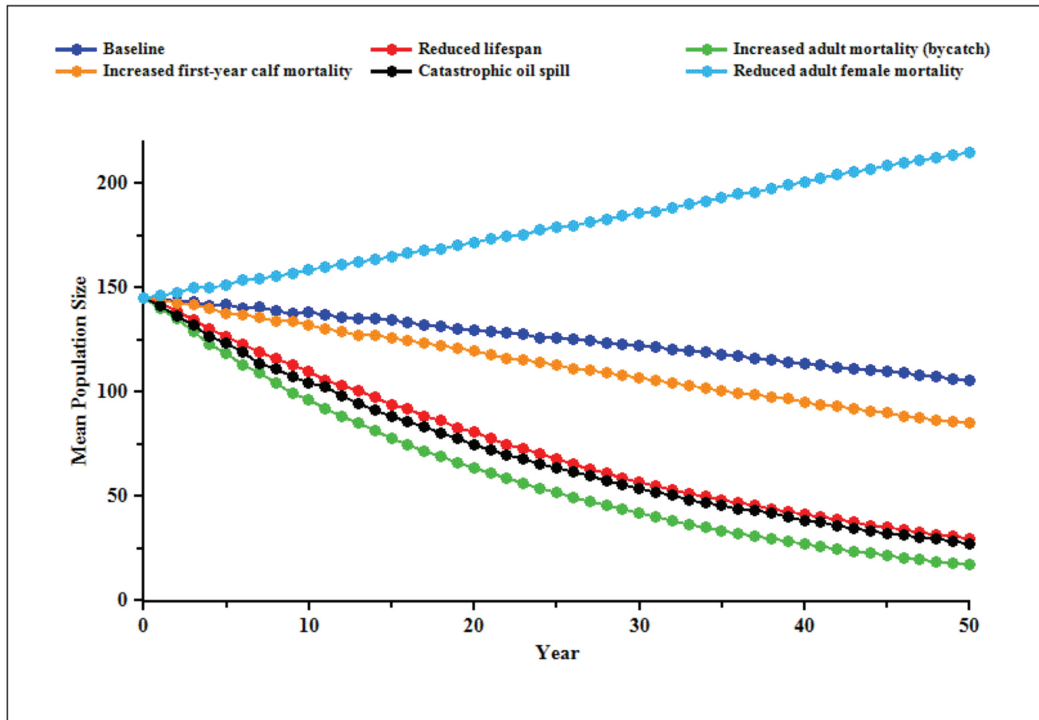
Figure 2. Mean population trajectories (1,000 iterations) of baseline and hypothetical scenario models over a 50-y projection in *Vortex*

Table 6. *Vortex* output values for baseline and hypothetical scenario models. r = stochastic growth rate; PE = probability of extinction (proportion of iterations in which the population went extinct); N-Extant = final population size; and SD = standard deviation.

Model	r	SD(r)	PE	N-Extant	SD(N-Extant)
Baseline	-0.007	0.03	0.000	105.33	23.38
Reduced lifespan	-0.0334	0.0485	0.000	29.62	11.33
Increased adult mortality (bycatch)	-0.0447	0.0628	0.012	17.34	7.47
Increased 1st-year calf mortality	-0.0113	0.0312	0.000	84.81	20.10
Catastrophic oil spill	-0.0451	0.1811	0.152	31.94	31.13
Reduced female mortality	0.0120	0.0074	0.000	213.05	41.10

estimate of 136 in 2015 was due to the fact that, although just 135 individual dolphins were identified and included in the IWDG catalogue in 2015, 10 known individuals not recorded that year were resighted again in 2016 (Baker et al., 2018a). Overall, these results indicate that mark-recapture is a reliable technique to estimate animal abundance, with precision and accuracy increasing as sampling effort increases (White et al., 1982).

Baker et al. (2018a) presented a discovery curve that reached a clear plateau in 2015 when no new individuals were identified. This suggested that all individuals in the population had been recorded. As no new individuals were found in the NPWS photo-identification for that year, the suggestion that the whole population may have been photographed and identified is supported. However, 95% CIs around the estimate from the mark-recapture model report that a maximum of 202 individuals may occur within the site and suggested that there might be still more individuals to be identified. It is unlikely that this number of

individuals have yet to be captured and, thus, is more likely a legacy of the model.

Interestingly, the most recent abundance estimate from the Shannon Estuary in 2018 using mark-recapture techniques (Rogan et al., 2018) produced a very similar abundance (139 ± 15.23; CV = 0.11; 95% CI = 121 to 160) to that calculated using the IWDG photo-identification catalogue in 2015 (Figure 3; Table 4), Direct comparisons are more difficult since Rogan et al. (2018) was carried out 3 y later, but a review of this new NPWS photo-identification catalogue from 2018 would be of interest to see if there are many new individuals not included in the IWDG catalogue. All abundance estimates from the Shannon Estuary between 1997 and 2018 (Ingram, 2000; Ingram & Rogan, 2003; Englund et al., 2007, 2008; Berrow et al., 2012; Rogan et al., 2015, 2018), including those presented herein, are of the same magnitude, suggesting a stable trend in the population over the past two decades (Figure 3; Table 4). The extant population (145) in 2015 (Baker et al., 2018a), although slightly higher than some estimates, falls within the CIs (Figure 3; Table 4) and is, without doubt, the most accurate abundance estimate published to date.

Comparing Monitoring Strategies
Higher survey efforts tend to increase the quality of data available which leads to greater modelling precision and accuracy. The ability of a triennial monitoring strategy in the Shannon Estuary to detect trends in population changes over reasonable timescales was explored by Englund et al. (2008). Bottlenose dolphins are classic k-strategists, with long lifespans and low reproductive rates (Leatherwood & Reeves, 1990). Therefore, monitoring programmes dedicated to this species generally require a large investment of time and resources (Wilson et al., 1999). One way to investigate the ability of a monitoring strategy to meet a required objective is to explore the power of a range of strategies (Gerrodette, 1987). Such an analysis was carried out on the bottlenose dolphin population in the Moray Firth (Wilson et al., 1999; Thompson et al., 2000) which is similar to the Shannon Estuary in terms of habitat type, latitude, research effort, and population size (Arso Civil et al., 2017, 2019; Cheney et al., 2019).

Power analysis was based on the knowledge of the growth rate (r) of the population and the coefficient of variation (CV) of an abundance estimate, and it enabled the number of regular survey periods until trend (r) detection to be determined.
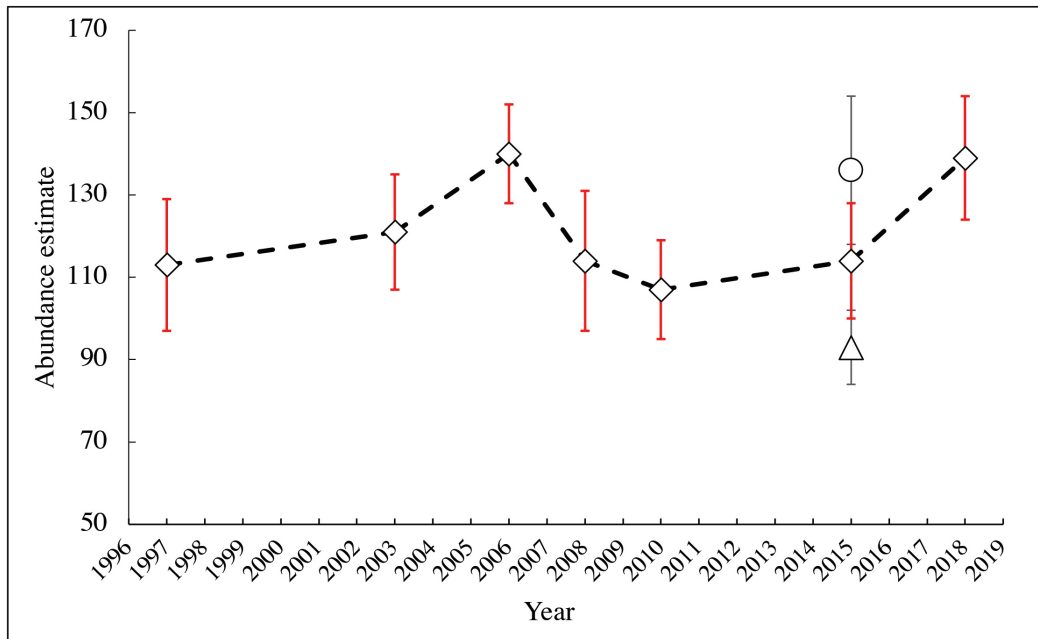

Figure 3. Mark-recapture abundance estimates (mean ± SE) of bottlenose dolphins in the Lower River Shannon Estuary SAC from 1997 to 2018, including those presented in this study: IWDG 2015 (circle) and revised NPWS 2015 (triangle). See Ingram (2000), Ingram & Rogan (2003), Englund et al. (2007, 2008), Berrow et al. (2012), and Rogan et al. (2015, 2018) for details.

This showed that a smaller number of periods between surveys was needed as growth rate was further from 0 (i.e., if growth slope is higher in absolute value) and the CV moves closer to 0 (i.e., the precision of the abundance estimate is higher). Thompson et al. (2000) suggested that precautionary management will always be more capable of detecting trends than traditional strategies (annual and quinquennial), but also that within traditional approaches, annual monitoring was more powerful than quinquennial monitoring. Englund et al. (2008) made a similar conclusion for the Shannon Estuary population and suggested that a triennial monitoring strategy was not as powerful for detecting trends in the population with the same number of annual survey periods unless very low CVs were achieved during each survey. Furthermore, some parameters relevant to understanding population trends cannot be easily obtained from a small number of surveys (12) conducted every 3 y. This is supported by Baker et al. (2018a) who carried out increased survey efforts between 2012 and 2015. Additionally, although photo-identification data obtained on a triennial basis can be used to calculate estimates of critical demographic parameters, this has not been carried out to date. NPWS triennial reports have focused on estimating abundance and distribution with less attention to other areas of study relevant to the long-lived and behaviourally complex bottlenose dolphin.

Population Viability Analysis
It is clear that a triennial monitoring strategy will be less able to detect population trends than annual surveys, but it will also be less able to provide accurate vital rates such as mortality and reproductive rates that are relevant to understanding these trends. Utilizing recent estimates of key parameters provided by Baker at al. (2018a), we present the most accurate PVA to date on the bottlenose dolphin population in the Shannon Estuary. There are few published PVA studies available on bottlenose dolphins (Thompson et al., 2000; Englund et al., 2008; Manlik et al., 2016), probably due to the need for long-term monitoring data and the economic and technical constraints. The baseline model showed a slowly declining population trend (Figure 2; Table 5), which is in contrast to previous abundance estimates carried out in the last 23 y, which showed a stable, if not increasing, population (Figure 3).

Sensitivity Analysis
Sensitivity analysis based on variation of key parameters within fixed proportions (±25% and ±50%) showed that adult female mortality followed by reproductive rates and lifespan were the most important parameters influencing population trajectories; whereas other parameters, such as adult male mortality, inbreeding, and initial population size, had minor influence. First-year calf mortality seemed to have an intermediate effect between these two groups of parameters. These results are consistent with Manlik et al.'s (2016) PVA for bottlenose dolphins in Bunbury (Western Australia); these dolphins showed higher sensitivity to adult mortality than calf mortality, reproduction rates, and inbreeding under fixed-proportion sensitivity analysis.

These results clearly demonstrate the importance of adult females and their reproductive capacity in population growth. Reducing both the proportion of females breeding and the distribution of broods per year by 50% led to a steep decline (r = -0.0317 and r = -0.0339, respectively). Reducing lifespan down to 20 y (-50%) drove the population to a quick depletion within 50 y of projection (r = -0.0877; PE = 0.7530). Furthermore, the sensitivity of population dynamics was higher when lifespan was reduced than when it was expanded (Table 5). Decreasing female mortality by 50% resulted in population growth (r = 0.0142); but when it was increased by 50%, the population declined (r = -0.0286). This illustrated that the fewer opportunities for reproduction available to females, the worse the population's fate would be. This was also consistent with the conclusions of Manlik et al. (2016) regarding the importance of reproduction in the viability of slow-growing populations (Manlik et al., 2019). In contrast, male influence on model output was minimal (Table 5), which is explained by the polygynous reproductive system typical of the species where females carry the burden of rearing offspring.

The relatively low importance of mortality of young calves is easily explained since calves are dependent on their mothers for a mean period of 3 y, and females cannot reproduce again during this period. If the calf dies before independence from the mother, she can reproduce again, and the lost calf is substituted. However, the cost of replacing adult breeding females if they die is much higher in terms of time since a female would need 10 y to reach sexual maturity. This also suggests that, even though low mortality of calves would be beneficial for population growth, it would be offset if female reproductive capacity was reduced either by high female mortality, reduced lifespan of the whole population, or reduced reproductive rates.

The Effect of Threat Scenarios on Population Viability
As a complementary approach for sensitivity analysis, several scenarios were simulated to help inform management.

Reduced Lifespan—Reducing expected lifespan from 50 to 30 y resulted in a clear population decline (r = -0.0334). Given this result and comparing it with the baseline model yielded by *Vortex* (Figure 2; Table 6) as well as the trend showed by the abundance estimates in the last decades (Figure 3), this scenario seems unrealistic. Nevertheless, along with sensitivity analysis, this result served to provide proof of the relevance of lifespan on population viability since the longer bottlenose dolphins live, the more chances they will have to mate and produce offspring.

A reduced lifespan like this one could not be driven by just one factor but by a combination of factors. Bioaccumulation of persistent pollutants, underwater noise, or low prey availability are just some of the threats taken on by coastal bottlenose dolphins around the world (Hammond et al., 2012), and the resultant cumulative effect may result in a low life expectancy, although this assertion is just speculation since there is no quantitative evidence on how these threats could affect marine mammal lifespans, and modelling of such a complex scenario is close to being a chimera—not just in the case of the Shannon population, but in the case of any marine mammal population around the globe—due to the lack of robust data. Thus, empirical data on the lifespan of bottlenose dolphins in Ireland through longitudinal studies or via studies using Growth Layer Groups in teeth (Sergeant et al., 1973; Hohn et al., 1989) would be of great interest in this case.

Increased Adult Mortality—An increase in adult mortality from 3.8 to 7%, which is equivalent to just two additional adults killed each year due to, for example, fisheries bycatch, had a negative effect on population trajectory (r = -0.0447). This result showed the sensitivity of the bottlenose population to moderate increases in adult mortality. Mortality causes remain largely unknown for this population, and the Shannon Estuary is not currently impacted by high fishing effort. However, these dolphins are known to occur outside the estuary (Ryan et al., 2010; Mirimin et al., 2011; Levesque et al., 2016; Nykänen et al., 2018) where they are more likely to encounter fisheries, such as gillnet fisheries, which are known to occasionally capture bottlenose dolphins (Berrow & Rogan, 1998). This simulation may help to understand the delicate state of this small dolphin population and the potential impact that even a small bycatch may have on its viability if fishing effort increased enough to produce such mortality in the future—whether within the estuary or in adjacent areas.

Increased First-Year Calf Mortality—Baker et al. (2018a) warned that current first-year calf mortality may be underestimated due to the small sample size used to calculate it. The exchange of a higher calf mortality value from the bottlenose dolphin population in Sarasota Bay (20%) provided by Wells & Scott (1990) yielded a slightly steeper declining trend compared with the baseline model (Figure 2; Table 6). Little is known about calf mortality in the Shannon Estuary. Some causes can be easily ruled out such as predation since there are no known predators of bottlenose dolphins in the estuary. Other causes may have an impact on calf survival but still remain largely unstudied. One of these possible threats is pollution. The Shannon Estuary is a major centre for Irish industry with several industrial facilities located along its shores (Figure 1). In addition, the River Shannon, which is the longest river in Ireland, and its tributary rivers catchment encompass large areas of farmland serving as potential additional sources of persistent pollutants.

Bottlenose dolphins in the Shannon Estuary are known to present high levels of PCB (Berrow et al., 2002) which, although found to be relatively lower than in other European populations (Jepson et al., 2016), are still close to, or occasionally above, current toxic thresholds. These legacy pollutants as well as new emerging pollutants could together affect calf viability (Wells et al., 2005; Murphy et al., 2010; Jepson & Law, 2016). Demographic effects of PCB have been explored in killer whales (Lacy et al., 2017) and bottlenose dolphins (Hall et al., 2006), demonstrating the negative long-term effects on young calf survival of the exchange of PCB between mothers and calves through gestation and lactation. Unfortunately, these kinds of models are much more complex than those presented herein and require more robust data without which their interpretation can be meaningless. Additional research effort is necessary to quantify current levels of persistent pollutants, including emerging pollutants, to test the potential effects of pollutants on population viability.

Catastrophic Oil Spill—A large oil spill with similar effects on reproduction and survival as the *Deepwater Horizon* oil spill which occurred in the Gulf of Mexico in 2010 (Lane et al., 2015; McDonald et al., 2017) was also simulated. If the event occurred with a frequency of 5%, which is equivalent to once every 20 y, it would drive the population to a steep decline (r = -0.0451), with a 15.2% probability of extinction within 50 y. This indicates that the population would be sensitive to sporadic catastrophic events, which significantly reduce reproduction and survival rates, even for a short period of time. Given the high number of large vessels using the Shannon Estuary, this is a real possibility. Prevention and mitigation measures are important, including risk assessment and modelling (Kirby & Law, 2010; Alves et al., 2015), and integrated post-incident contingency plans such as

the Shannon Estuary Anti-Pollution Team (SEA-PT) (2019).

Reduced Adult Female Mortality—Given that baseline adult mortality rate is a maximum estimate of real mortality (Baker et al., 2018a) and is not age- or sex-specific, a reduction in adult female mortality from 3.8 to 2.5% (saving just one female a year) was also modelled. This scenario led the population to a moderate positive growth (r = 0.012), which shows the great influence that female reproductive capacity has on the fate of the population. Knowing the probable inaccuracy of current mortality estimates and its influence on population viability, it is recommended that more effort be placed on providing a revised estimate, which may be lower than 3.8%. This scenario can serve as a reference since, if it is assumed that the population is stable—that is, the growth rate is approximately 0 (Figure 3)—this growth rate would be reached with an adult female mortality rate between 3.8 and 2.5%. Further analysis should be done to test this assumption.

General Considerations Arising from PVA—This study provides a good reference for both managers and researchers responsible for managing the conservation status of the bottlenose dolphin population in the Shannon Estuary. There is still a considerable lack of knowledge regarding some of the parameters used in our models. Nothing is known about their life expectancy or the age at which these bottlenose dolphins reach sexual maturity. Moreover, although the age proportions are well documented, gender is known for less than half (49%) of all identified individuals, and those that are known are biased towards females (82%) (Baker et al., 2018a).

Present mortality rates are just age-specific for first-year calves but not for the rest of the age classes or sex groups. Also, mean annual mortality rate for adults and juveniles of 3.8%, given by Baker et al. (2018a), is a maximum value and takes into account all individuals that have not been seen for several years, assuming they are dead. This may not be realistic since it is known that bottlenose dolphins in the Shannon Estuary present a typical fission-fusion social structure (Foley et al., 2010) and, thus, some individuals, particularly males, travel out of the estuary (Levesque et al., 2016). Mirimin et al. (2011) reported that dolphins in the Shannon Estuary were genetically related to a small group of dolphins which took up residency in Cork Harbour (Ryan et al., 2010), more than 300 km away from the Shannon Estuary, suggesting that there has been migration out of the estuary. Therefore, present mortality rates may be lower because some of the individuals presumed dead and included in calculating mortality estimates may be actually alive and

possibly seen again in the estuary in the future. The forecast of a declining population (Figure 2; Table 6) is in contrast to the small variation in abundance estimates reported over the last 23 y which showed an apparently stable population (Figure 3). Given the high sensitivity of models to this parameter and its probable inaccuracy, it is likely that the declining trend yielded by *Vortex* is strongly biased by values used for adult female mortality.

This study suggests that a detailed study on lifespan, sex distribution, and age- and sex-specific mortality rates should be carried out because many of these parameters are directly related to female reproductive capacity and availability. Future research should focus on gaining new and more accurate information on those parameters. This will enable a better understanding of the population's dynamics and provide more precise and accurate PVA models.

With regards to management, results suggested that great care and concern should be taken to avoid the loss of breeding females and the potential impacts of catastrophic events such as large oil spills. Moreover, it would be desirable, as suggested by Englund et al. (2008), to monitor fishing activity for both incidental capture and prey removal within the estuary and adjacent areas as well as to control and monitor the presence of persistent pollutants such as PCB and the intense industrial activity within the Shannon Estuary. Such actions would assist in maintaining the bottlenose dolphin population in the Shannon Estuary at Favourable Conservation Status (Epstein et al., 2016), as well as to reach its related attributes and targets to meet the obligations established by current European legislation.

Literature Cited

Akçakaya, H. R., & Sjögren-Gulve, P. (2000). Population viability analyses in conservation planning: An overview. *Ecological Bulletins*, *48*, 9-21.

Allen, S. J., Pollock, K. H., Bouchet, P. J., Kobryn, H. T., McElligott, D. B., Nicholson, K. E., Smith, J. N., &

Loneragan, N. R. (2017). Preliminary estimates of the abundance and fidelity of dolphins associating with a demersal trawl fishery. *Scientific Reports*, *7*, 4995. https://doi.org/10.1038/s41598-017-05189-0

Alves, T. M., Kokinou, E., Zodiatis, G., Lardner, R., Panagiotakis, C., & Radhakrishnan, H. (2015). Modelling of oil spills in confined maritime basins: The case for early response in the Eastern Mediterranean Sea. *Environmental Pollution*, *206*, 390-399. https://doi.org/10.1016/j.envpol.2015.07.042

Arso Civil, M., Cheney, B., Quick, N. J., Thompson, P. M., & Hammond, P. S. (2017). A new approach to estimate fecundity rate from inter-birth intervals. *Ecosphere*, *8*(4), e01796. https://doi.org/10.1002/ecs2.1796

Arso Civil, M., Cheney, B., Quick, N. J., Islas-Villanueva, V., Graves, J. A., Janik, V. M., Thompson, P. M., & Hammond, P. S. (2019). Variations in age- and sex-specific survival rates help explain population trend in a discrete marine mammal population. *Ecology and Evolution*, *9*(1), 533-544. https://doi.org/10.1002/ece3.4772

Baker, I., O'Brien, J., McHugh, K., & Berrow, S. (2018a). Female reproductive parameters and population demographics of bottlenose dolphins (*Tursiops truncatus*) in the Shannon Estuary, Ireland. *Marine Biology*, *165*(1), 15. https://doi.org/10.1007/s00227-017-3265-z

Baker, I., O'Brien, J., McHugh, K., Ingram, S. N., & Berrow, S. (2018b). Bottlenose dolphin (*Tursiops truncatus*) social structure in the Shannon Estuary, Ireland, is distinguished by age- and area-related associations. *Marine Mammal Science*, *34*(2), 458-487. https://doi.org/10.1111/mms.12462

Barlow, J., Calambokidis, J., Falcone, E. A., Baker, C. S., Burdin, A. M., Clapham, P. J., Ford, J. K. B., Gabriele, C. M., LeDuc, R., Mattila, D. K., Quinn II, T. J., Rojas-Bracho, L., Straley, J. M., Taylor, B. L., Urbán R., J., Wade, P., Weller, D., Witteveen, B. H., & Yamaguchi, M. (2011). Humpback whale abundance in the North Pacific estimated by photographic capture-recapture with bias correction from simulation studies. *Marine Mammal Science*, *27*(4), 793-818. https://doi.org/10.1111/j.1748-7692.2010.00444.x

Berrow, S. D. (2009). Winter distribution of bottle-nosed dolphins (*Tursiops truncatus* (Montagu)) in the inner Shannon Estuary. *The Irish Naturalists' Journal*, *30*(1), 35-39.

Berrow, S. D., & Rogan, E. (1998). Incidental capture of cetaceans in Irish waters. *The Irish Naturalists' Journal*, *26*(1/2), 22-31.

Berrow, S. D., Holmes, B., & Kiely, O. R. (1996). Distribution and abundance of bottle-nosed dolphins *Tursiops truncatus* (Montagu) in the Shannon Estuary. *Biology and Environment: Proceedings of the Royal Irish Academy*, *96*(1), 1-9.

Berrow, S., O'Brien, J., & Baker, I. (2016, December). *The Shannon population: Status and situation management*. Proceedings of the Translational Bottlenose Dolphin Conservation Workshop, Dublin, Ireland.

Berrow, S., O'Brien, J., Groth, L., Foley, A., & Voigt, K. (2012). Abundance estimate of bottlenose dolphins (*Tursiops truncatus*) in the Lower River Shannon candidate Special Area of Conservation, Ireland. *Aquatic Mammals*, *38*(2), 136-144. https://doi.org/10.1578/AM.38.2.2012.136

Berrow, S. D., McHugh, B., Glynn, D., McGovern, E., Parsons, K. M., Baird, R. W., & Hooker, S. K. (2002). Organochlorine concentrations in resident bottlenose dolphins (*Tursiops truncatus*) in the Shannon Estuary, Ireland. *Marine Pollution Bulletin*, *44*(11), 1296-1303. https://doi.org/10.1016/S0025-326X(02)00215-1

Cheney, B. J., Thompson, P. M., & Cordes, L. S. (2019). Increasing trends in fecundity and calf survival of bottlenose dolphins in a marine protected area. *Scientific Reports*, *9*(1), 1767. https://doi.org/10.1038/s41598-018-38278-9

Dos Santos, M. E., & Lacerda, M. (1987). Preliminary observations of the bottlenose dolphin (*Tursiops truncatus*) in the Sado Estuary (Portugal). *Aquatic Mammals*, *13*(2), 65-80.

Englund, A., Ingram, S., & Rogan, E. (2007). *Population status report for bottlenose dolphins using the Lower River Shannon SAC, 2006-2007*. Final Report to the National Parks and Wildlife Service.

Englund, A., Ingram, S., & Rogan, E. (2008). *An updated population status report for bottlenose dolphins using the Lower River Shannon SAC in 2008*. Report to the National Parks and Wildlife Service.

Epstein, Y., López-Bao, J. V., & Chapron, G. (2016). A legal-ecological understanding of favorable conservation status for species in Europe. *Conservation Letters*, *9*(2), 81-88. https://doi.org/10.1111/conl.12200

Evans, P. G. H., Anderwald, P., & Baines, M. E. (2003). *UK cetacean status review*. Report to English Nature and Countryside Council for Wales.

Fantle-Lepczyk, J., Berry, L., Lepczyk, C., Duffy, D., & Conant, S. (2018a). Key demographic factors for recovery of the endangered nightingale reed-warbler (*Acrocephalus hiwae*) via population viability analysis. *Avian Conservation and Ecology*, *13*(2), 13. https://doi.org/10.5751/ACE-01289-130213

Fantle-Lepczyk, J., Taylor, A., Duffy, D. C., Crampton, L. H., & Conant, S. (2018b). Using population viability analysis to evaluate management activities for an endangered Hawaiian endemic, the Puaiohi (*Myadestes palmeri*). *PLOS ONE*, *13*(6), e0198952. https://doi.org/10.1371/journal.pone.0198952

Fletcher, W. J., & Santoro, K. (Eds.). (2012). *Status reports of the fisheries and aquatic resources of Western Australia 2012/13: The state of fisheries*. Department of Fisheries.

Foley, A., McGrath, D., Berrow, S., & Gerritsen, H. D. (2010). Social structure within the bottlenose dolphin (*Tursiops truncatus*) population in the Shannon Estuary, Ireland. *Aquatic Mammals*, *36*(4), 372-381. https://doi.org/10.1578/AM.36.4.2010.372

Frasier, T. R., Hamilton, P. K., Brown, M. W., Kraus, S. D., & White, B. N. (2009). Sources and rates of errors in methods of individual identification for North Atlantic right whales. *Journal of Mammalogy*, *90*(5), 1246-1255. https://doi.org/10.1644/08-MAMM-A-328.1

Friday, N. T. D., Smith, P. T., Stevick, J. A., & Fernald, T. (2008). Balancing bias and precision in capture-recapture estimates of abundance. *Marine Mammal Science*, *24*, 253-275. https://doi.org/10.1111/j.1748-7692.2008.00187.x

Gerritsen, H. D., & Lordan, C. (2014). *Atlas of commercial fisheries around Ireland*. Marine Institute. ISBN 978-1-902895-56-7

Gerrodette, T. (1987). A power analysis for detecting trends. *Ecology*, *68*(5), 1364-1372. https://doi.org/10.2307/1939220

Gong, M., Song, Y., Yang, Z., & Lin, C. (2012). Important population viability analysis parameters for giant pandas (*Aliuropoda melanoleuca*). *Zoological Research*, *33*(E1-2), 18-24. https://doi.org/10.3724/SP.J.1141.2012.E01-02E18

Gunnlaugsson, T., & Sigurjónsson, J. (1990). A note on the problem of false positives in the use of natural marking data for abundance estimation. In P. S. Hammond, S. A. Mizroch, & G. P. Donovan (Eds.), *Individual recognition of cetaceans: Use of photo-identification and other techniques to estimate population parameters*. *Report of the International Whaling Commission*, Special Issue 12, 143-145.

Hall, A. J., McConnell, B. J., Rowles, T. K., Aguilar, A., Borrell, A., Schwacke, L., Reijnders, P. J. H., & Wells, R. S. (2006). Individual-based model framework to assess population consequences of polychlorinated biphenyl exposure in bottlenose dolphins. *Environmental Health Perspectives*, *114*(Supp. 1), 60-64.

Hammond, P. S., Bearzi, G., Bjørge, A., Forney, K. A., Karczmarski, L., Kasuya, T., Perrin, W. F., Scott, M. D., Wang, J. Y., Wells, R. S., & Wilson, B. (2012). *Tursiops truncatus*. In International Union for Conservation of Nature (Ed.), *The IUCN red list of threatened species 2012* (e.T22563A17347397). https://doi.org/10.2305/IUCN.UK.2012.RLTS.T22563A17347397.en

Hohn, A. A., Scott, M. D., Wells, R. S., Sweeney, J. C., & Irvine, A. B. (1989). Growth layers in teeth from known-age, free-ranging bottlenose dolphins. *Marine Mammal Science*, *5*(4), 315-342. https://doi.org/10.1111/j.1748-7692.1989.tb00346.x

Ingram, S. N. (2000). *The ecology and conservation of bottlenose dolphins in the Shannon Estuary, Ireland* (Unpub. doctoral dissertation). University College Cork, Corcaigh, Ireland.

Ingram, S. N., & Rogan, E. (2002). Identifying critical areas and habitat preferences of bottlenose dolphins *Tursiops truncatus*. *Marine Ecology Progress Series*, *244*, 247-255. https://doi.org/10.3354/meps244247

Ingram, S., & Rogan, E. (2003). *Bottlenose dolphins (*Tursiops truncatus*) in the Shannon Estuary and selected areas of the west-coast of Ireland*. Report to the National Parks and Wildlife Service.

Irvine, B. (2019). *How long do dolphins live?* www.sarasotadolphin.org/how-long-do-dolphins-live-2

Jaakkola, K., & Willis, K. (2019). How long do dolphins live? Survival rates and life expectancies for bottlenose dolphins in zoological facilities vs. wild populations. *Marine Mammal Science*, *35*(4), 1418-1437. https://doi.org/10.1111/mms.12601

Jepson, P. D., & Law, R. J. (2016). Persistent pollutants, persistent threats. *Science*, *352*(6292), 1388-1389. https://doi.org/10.1126/science.aaf9075

Jepson, P. D., Deaville, R., Barber, J. L., Aguilar, À., Borrell, A., Murphy, S., Barry, J., Brownlow, A., Barnett, J., Berrow, S., Cunningham, A. A., Davison, N. J., ten Doeschate, M., Esteban, R., Ferreira, M., Foote, A. D., Genov, T., Giménez, J., Loveridge, J., Llavona, Á., . . . Law, R. J. (2016). PCB pollution continues to impact populations of orcas and other dolphins in European waters. *Scientific Reports*, *6*, 18573. https://doi.org/10.1038/srep18573

Kirby, M. F., & Law, R. J. (2010). Accidental spills at sea—Risk, impact, mitigation and the need for co-ordinated post-incident monitoring. *Marine Pollution Bulletin*, *60*(6), 797-803. https://doi.org/10.1016/j.marpolbul.2010.03.015

Kiszka, J., Hassani, S., & Pezeril, S. (2004). Distribution and status of small cetaceans along the French channel coasts: Using opportunistic records for a preliminary assessment. *Lutra*, *47*(1), 33-46.

Lacy, R. C. (1993). *Vortex*: A computer simulation model for population viability analysis. *Wildlife Research*, *20*(1), 45-65. https://doi.org/10.1071/WR9930045

Lacy, R. C. (2019). Lessons from 30 years of population viability analysis of wildlife populations. *Zoo Biology*, *38*, 67-77. https://doi.org/10.1002/zoo.21468

Lacy, R. C., Williams, R., Ashe, E., Balcomb III, K. C., Brent, L. J., Clark, C. W., Croft, D. P., Giles, D. A., MacDuffee, M., & Paquet, P. C. (2017). Evaluating anthropogenic threats to endangered killer whales to inform effective recovery plans. *Scientific Reports*, *7*(1), 1-12. https://doi.org/10.1038/s41598-017-14471-0

Lane, S. M., Smith, C. R., Mitchell, J., Balmer, B. C., Barry, K. P., McDonald, T., Mori, C. S., Rosel, P. E., Rowles, T. K., Speakman, T. R., Townsend, F. I., Tumlin, M. C., Wells, R. S., Zolman, E. S., & Schwacke, L. H. (2015). Reproductive outcome and survival of common bottlenose dolphins sampled in Barataria Bay, Louisiana, USA, following the *Deepwater Horizon* oil spill. *Proceedings of the Royal Society B: Biological Sciences*, *282*, 20151944. https://doi.org/10.1098/rspb.2015.1944

Leatherwood, S., & Reeves, R. R. (1983). *The Sierra Club handbook of whales and dolphins*. Sierra Club Books.

Leatherwood, S., & Reeves, R. R. (Eds.). (1990). *The bottlenose dolphin*. Academic Press.

Levesque, S., Reusch, K., Baker, I., O'Brien, J., & Berrow, S. (2016). Photo-identification of bottlenose dolphins (*Tursiops truncatus*) in Tralee Bay and Brandon Bay, Co. Kerry: A case for SAC boundary extension. *Biology and Environment: Proceedings of the Royal Irish Academy*, *116*(2), 109-118. https://doi.org/10.3318/bioe.2016.11

Limerick City and County Council. (2019). *Strategic integrated framework plan for the Shannon Estuary (2013-*

2020). RPS Consultants. www.limerick.ie/council/strate-gic-integrated-framework-plan

Manlik, O. (2019). The importance of reproduction for the conservation of slow-growing animal populations. In P. Comizzoli, J. Brown, & W. V. Holt (Eds.), *Advances in experimental medicine and biology: Vol. 1200. Reproductive science in animal conservation—Progress and prospects* (2nd ed., pp. 13-39). Springer. https://doi.org/10.1007/978-3-030-23633-5_2

Manlik, O., Lacy, R. C., & Sherwin, W. B. (2018). Applicability and limitations of sensitivity analyses for wildlife management. *Journal of Applied Ecology*, *55*, 1430-1440. https://doi.org/10.1111/1365-2664.13044

Manlik, O., McDonald, J. A., Mann, J., Raudino, H. C., Bejder, L., Krützen, M., Connor, R. C., Heithaus, M. R., Lacy, R. C., & Sherwin, W. B. (2016). The relative importance of reproduction and survival for the conservation of two dolphin populations. *Ecology and Evolution*, *6*(11), 3496-3512. https://doi.org/10.1002/ece3.2130

McDonald, T. L., Hornsby, F. E., Speakman, T. R., Zolman, E. S., Mullin, K. D., Sinclair, C., Rosel, P. E., Thomas, L., & Schwacke, L. H. (2017). Survival, density, and abundance of common bottlenose dolphins in Barataria Bay (USA) following the *Deepwater Horizon* oil spill. *Endangered Species Research*, *33*, 193-209. https://doi.org/10.3354/esr00806

Methion, S., & Díaz López, B. (2018). Abundance and demographic parameters of bottlenose dolphins in a highly affected coastal ecosystem. *Marine and Freshwater Research*, *69*(9), 1355-1364. https://doi.org/10.1071/MF17346

Mills, L. S., & Lindberg, M. S. (2002). Sensitivity analysis to evaluate the consequences of conservation action. In S. R. Beissinger & D. R. McCullough (Eds.), *Population viability analysis* (pp. 338-366). University of Chicago Press.

Mirimin, L., Miller, R., Dillane, E., Berrow, S. D., Ingram, S., Cross, T. F., & Rogan, E. (2011). Fine-scale population genetic structuring of bottlenose dolphins in Irish coastal waters. *Animal Conservation*, *14*(4), 342-353. https://doi.org/10.1111/j.1469-1795.2010.00432.x

Morizur, Y., Berrow, S. D., Tregenza, N. J. C., Couperus, A. S., & Pouvreau, S. (1999). Incidental catches of marine-mammals in pelagic trawl fisheries of the northeast Atlantic. *Fisheries Research*, *41*(3), 297-307. https://doi.org/10.1016/S0165-7836(99)00013-2

Morris, W. F., & Doak, D. F. (2002). *Quantitative conservation biology*. Sinauer.

Murphy, S., Pierce, G. J., Law, R. J., Bersuder, P., Jepson, P. D., Learmonth, J. A., Addink, M., Dabin, W., Santos, M. B., Deaville, R., Zegers, B. N., Mets, A., Rogan, E., Ridoux, V., Reid, R. J., Smeenk, C., Jauniaux, T., López, A., Alonso Farré, J. M., González, A. F., . . . Boon, J. P. (2010). Assessing the effect of persistent organic pollutants on reproductive activity in common dolphins and harbour porpoises. *Journal of Northwest Atlantic*

Fishery Science, *4*, 153-173. https://doi.org/10.2960/J.v42.m658

National Parks and Wildlife Service (NPWS). (2012). *Lower River Shannon SAC (site code: 2165): Conservation objectives supporting document – Marine habitats and species*. NPWS, Department of Arts, Heritage and the Gaeltacht.

Nykänen, M., Dillane, E., Englund, A., Foote, A. D., Ingram, S. N., Louis, M., Mirimin, L., Oudejans, M., & Rogan, E. (2018). Quantifying dispersal between marine protected areas by a highly mobile species, the bottlenose dolphin, *Tursiops truncatus*. *Ecology and Evolution*, *8*(18), 9241-9258. https://doi.org/10.1002/ece3.4343

O'Brien, J. M., Beck, S., Berrow, S. D., André, M., van der Schaar, M., O'Connor, I., & McKeown, E. P. (2016). The use of deep-water berths and the effect of noise on bottlenose dolphins in the Shannon Estuary cSAC. In A. N. Popper & A. Hawkins (Eds.), *Advances in experimental medicine and biology: Vol. 875. The effects of noise on aquatic life II* (pp. 775-783). Springer. https://doi.org/10.1007/978-1-4939-2981-8_95

O'Grady, O., Brook, B. W., Reed, D. H., Ballou, J. D., Tonkyn, D. W., & Frankham, R. (2006). Realistic levels of inbreeding depression strongly affect extinction risk in wild populations. *Biological Conservation*, *133*, 42-51. https://doi.org/10.1016/j.biocon.2006.05.016

Otis, D. L., Burnham, K. P., White, G. C., & Anderson, D. R. (1978). Statistical inference from capture data on closed animal populations. *Wildlife Monographs*, *62*, 3-135.

Pulliam, H. R., Dunning, J. B., Jr., & Liu, J. (1992). Population dynamics in complex landscapes: A case study. *Ecological Applications*, *2*(2), 165-177. https://doi.org/10.2307/1941773

Read, A. J., Wells, R. S., Hohn, A. A., & Scott, M. D. (1993). Patterns of growth in wild bottlenose dolphins, *Tursiops truncatus*. *Journal of Zoology*, *231*(1), 107-123. https://doi.org/10.1111/j.1469-7998.1993.tb05356.x

Rexsted, E., & Burnham, K. P. (1991). *User's guide for interactive program* CAPTURE. Colorado Cooperative Fish and Wildlife Research Unit.

Rogan, E., Ingram, S., Holmes, B., & O'Flanagan, C. (2000). *A survey of bottlenose dolphins (*Tursiops truncatus*) in the Shannon Estuary*. Marine Institute.

Rogan, E., Kelly, T., Ingram, S., & Roycroft, D. (2003). *The ecology of seabirds and marine mammals in a fluctuating marine environment*. Report to the Higher Education Authority.

Rogan, E., Nykänen, M., Gkaragkouni, M., & Ingram, S. (2015). *Bottlenose dolphin surveys in the Lower River Shannon SAC, 2015*. Final Report to the National Parks and Wildlife Service, the Department of Arts, Heritage and the Gaeltacht.

Rogan, E., Garagouni, M., Nykänen, M., Whitaker, A., & Ingram, S. N. (2018). *Bottlenose dolphin surveys in the Lower River Shannon SAC, 2018*. Final Report to the

National Parks and Wildlife Service, the Department of Arts, Heritage and the Gaeltacht.

Ryan, C., Rogan, E., & Cross, T. (2010). The use of Cork Harbour by bottlenose dolphins (*Tursiops truncatus* (Montagu, 1821)). *The Irish Naturalists' Journal*, *31*(1), 1-9.

Shannon Estuary Anti-Pollution Team (SEA-PT). (2019). *Shannon Estuary oil contingency plan*. www.seapt.ie/wp-content/uploads/2012/04/Shannon-Estuary-OSCP_Final.pdf

Shannon Foynes Port Company (SFPC). (2013). *Shannon Foynes Port Company – Vision 2041*. https://www.sfpc.ie/download/SFPC%20MASTERPLAN%20Final.pdf

Sergeant, D. E., Caldwell, D. K., & Caldwell, M. C. (1973). Age, growth, and maturity of bottle-nosed dolphin (*Tursiops truncatus*) from northeast Florida. *Journal of the Fisheries Board of Canada*, *30*(7), 1009-1011. https://doi.org/10.1139/f73-165

Stevick, P. T., Palsbøll, P. J., Smith, T. D., Bravington, M. V., & Hammond, P. S. (2001). Errors in identification using natural markings: Rates, sources, and effects on capture-recapture estimates of abundance. *Canadian Journal of Fisheries and Aquatic Sciences*, *58*(9), 1861-1870. https://doi.org/10.1139/f01-131

Thompson, P. M., Wilson, B., Grellier, K., & Hammond, P. S. (2000). Combining power analysis and population viability analysis to compare traditional and precautionary approaches to conservation of coastal cetaceans. *Conservation Biology*, *14*(5), 1253-1263. https://doi.org/10.1046/j.1523-1739.2000.00099-410.x

Urian, K., Gorgone, A., Read, A., Balmer, B., Wells, R. S., Berggren, P., Durban, J., Eguchi, T., Rayment, W., & Hammond, P. S. (2015). Recommendations for photo-identification methods used in capture-recapture models with cetaceans. *Marine Mammal Science*, *31*(1), 298-321. https://doi.org/10.1111/mms.12141

Wakefield, C. B., Blight, S., Dorman, S. R., Denham, A., Newman, S. J., Wakeford, J., Molony, B. W., Thomson, A. W., Syers, C., & O'Donoghue, S. (2014). *Independent observations of catches and subsurface mitigation efficiencies of modified trawl nets for endangered, threatened and protected megafauna bycatch in the Pilbara Fish Trawl Fishery* (Fisheries Research Report No. 244). Department of Fisheries.

Wells, R. S., & Scott, M. D. (1990). Estimating bottlenose dolphin population parameters from individual identification and capture-release techniques. *Report of the International Whaling Commission*, Special Issue 12, 407-415.

Wells, R. S., & Scott, M. D. (1999). Bottlenose dolphin *Tursiops truncatus* (Montagu, 1821). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of marine mammals: The second book of dolphins and porpoises* (Vol. 6, pp. 137-182). Academic Press.

Wells, R. S., Tornero, V., Borrell, A., Aguilar, A., Rowles, T. K., Rhinehart, H. L., Hofmann, S., Jarman, W. M., Hohn, A. A., & Sweeney, J. C. (2005). Integrating life-history and reproductive success data to examine potential relationships with organochlorine compounds for bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. *Science of the Total Environment*, *349*(1-3), 106-119. https://doi.org/10.1016/j.scitotenv.2005.01.010

White, G. C., Anderson, D. R., Burnham, K. P., & Otis, D. L. (1982). *Capture-recapture and removal methods for sampling closed populations*. Los Álamos National Laboratory.

Wilson, B., Hammond, P. S., & Thompson, P. M. (1999). Estimating size and assessing trends in a coastal bottlenose dolphin population. *Ecological Applications*, *9*(1), 288-300. https://doi.org/10.1890/1051-0761(1999)009[0288:ESAATI]2.0.CO;2

Würsig, B., & Würsig, M. (1977). The photographic determination of group size, composition, and stability of coastal porpoises (*Tursiops truncatus*). *Science*, *198*(4318), 755-756. https://doi.org/10.1126/science.198.4318.755

Yoshizaki, J., Pollock, K. H., Brownie, C., & Webster, R. A. (2009). Modeling misidentification errors in capture-recapture studies using photographic identification of evolving marks. *Ecology*, *90*(1), 3-9. https://doi.org/10.1890/08-0304.1