Update on the population and spatial ecology of bottlenose dolphins in the Bay of Islands

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Caption for title image. Terehu in the Bay of Islands. [Jochen Zaeschmar, Far Out Ocean Research Collective]

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Background: The Bay of Islands provides a key habitat for bottlenose dolphins from the north-eastern New Zealand population, with the area having historically high numbers of semi-resident dolphins, and frequent occurrence of transient individuals (hereafter the 'local population'). In 2021, the Department of Conservation and a collective of partners from Te Pēwhairangi hapū established Te Pēwhairangi/Bay of Islands Marine Mammal Sanctuary, that aimed to limit vessel-related impacts on the local marine mammals, including bottlenose dolphins. There has been limited analysis focussing on the demographic parameters of bottlenose dolphins in the area since 2013. Thus, critical information for the management of the species in the Bay of Islands is outdated. In this study, we update demographic parameters and information on the spatial ecology of bottlenose dolphins using the Bay of Islands by: 1) integrating the full breadth of data from multiple research providers to establish a continuous, long-term database; 2) carrying out fieldwork to provide new information on population ecology; and 3) using the combined long-term and recent datasets to calculate updated estimates of local population abundance, survival, temporary emigration, calf mortality, reproductive rate and spatial distribution patterns. We also carried out an assessment of compliance with the sanctuary rules to determine the effectiveness of management for limiting the impacts of vessels on marine mammals.

Approach: A vessel-based fieldwork programme was initiated in March 2024, with daily surveys of the Bay of Islands carried out until the end of May 2024. Surveys utilised visual survey methods, photo-identification and acoustic deployments to gather new data on the local population. Data from multiple institutions were critically reviewed and merged to create three master databases on 1) dolphin encounters, 2) photo-identification of individual dolphins, and 3) reproductive parameters. The long-term photo-identification database provided input data for mark-recapture models using the robust design framework to update the abundance, survival and emigration rates of the local population. Using the dolphin encounter database, dynamic species distribution models were developed to determine the spatiotemporal distribution and habitat use of dolphins in the Bay of Islands over three decades and over seasonal scales. The reproduction database was used to generate updated, annual estimates of calf mortality and reproductive rate.

Results: Twenty days of fieldwork were undertaken between March and June 2024 during which twenty-four individual dolphins were photo-identified. Compliance with the sanctuary rules was generally poor, with power driven vessels rarely adhering to rules minimising dolphin interactions or the safe zones. However, all vessels were compliant with rules preventing swimming with marine mammals. Acoustic monitoring confirmed that dolphins were rarely present within the safe zones designated as part of the sanctuary. Using historical photo-ID data, over 128,000 images of Bay of Islands' dolphins were cross-matched, quality appraised and reviewed to establish a long-term database for demographic analysis. The final database contains records for 540 unique individuals over 1,605 daily surveys between 1994 and 2024. In line with previous findings, robust design analysis suggested the continuation of a trend of significant decline in local abundance, from a high point of 244 individuals (95%CI = 116–515) in the summer of 1997 and reaching its lowest point of 16 individuals (95% confidence interval = 16 – 16) in Summer 2019. Local abundance for Autumn of 2024 was estimated at 40 individuals (95% confidence interval = 27 – 58). Calf mortality has been variable over time but remains high compared to other areas. However, variability in field effort may bias calf mortality estimates. Temporally dynamic spatial models indicate seasonal changes in distribution, with lower occurrence during winter months. Areas of importance for the dolphins in the Bay of Islands have remained relatively stable over time, with important areas having limited overlap with safe zones implemented under the BOIMMS, while the general suitability of habitat has increased.

Conclusions: Multiple periods of decline in local abundance and high calf mortality in bottlenose dolphins in the Bay of Islands merits management intervention. While data on the compliance with the BOIMMS presented in this study has limitations due to the relatively short study duration (20 days), the available information suggests the current sanctuary may be limited by a lack of compliance and low dolphin occurrence within areas of heightened protection. If such low rates of compliance occur throughout the year, refining the existing sanctuary rules, heightened enforcement or considering alternative management approaches are recommended. The long-term databases generated in this study provide unique opportunities to guide management with targeted research into mechanisms behind the declines in local abundance, and to explore stakeholder-driven marine spatial planning scenarios that may improve the effectiveness of management of this important area.

1 Background

Common bottlenose dolphins (*Tursiops truncatus*/terehu) are a threatened species in Aotearoa New Zealand (NZ) (Baker et al. 2019) and are of significant cultural, recreational, commercial and ecological value. In NZ, bottlenose dolphins are represented by two ecotypes: an offshore form that typically inhabits waters greater than 100 m deep and can be found throughout New Zealand waters (Zaeschmar et al. 2020) and a coastal form that inhabits shallow coastal waters less than approximately 100 m deep (Constantine et al. 2004, Currey et al. 2009, Merriman et al. 2009). Regional populations of coastal bottlenose dolphins are found along the north-east coast of the North Island (Constantine et al. 2004), in the Marlborough Sounds (Merriman et al. 2009), Fiordland (Williams et al. 1993) and in the southernmost part of the South Island (Brough et al. 2015). Three of these populations are genetically different from each other (north-east coast of the North Island, Marlborough Sounds and Fiordland) with the fourth not yet evaluated (Tezanos-Pinto et al, 2009). Long term research on local, semi-resident populations has focussed on a few regions such as Fiordland (Currey et al. 2009, Bennington et al. 2023) and the Bay of Islands (Tezanos-Pinto et al. 2013). In both locations, local populations have a history of decline and variability in demographic rates (Currey et al. 2007, Currey et al. 2011, Hamilton 2013, Tezanos-Pinto et al. 2013, Tezanos-Pinto et al. 2015), with impacts associated with tourism (Lusseau 2003, Constantine et al. 2004) and habitat modification (in Fiordland) (Currey et al. 2011) being put forward as contributors to population variability.

The Bay of Islands provides important habitat for the wide-ranging north-eastern New Zealand population with the area having historically high numbers of semi-resident dolphins, and frequent occurrence of transient individuals (hereafter the 'local population'). Given the history of local population decline (Hamilton 2013, Tezanos-Pinto et al. 2013), high calf mortality (Tezanos-Pinto et al. 2015) and the impacts of tourism on dolphin behaviour (Constantine et al. 2004, Peters & Stockin 2016, Peters 2018), monitoring of the local population has been undertaken on behalf of the Department of Conservation (DOC) for three decades. This research aims to provide the best available information on population demographics (i.e., abundance, survival, reproductive rate, emigration) and spatial ecology (i.e., distribution/habitat use) to guide the management of the local population. In 2021, updated information on the low numbers of dolphins using the Bay of Islands, high apparent calf mortality and continuing impacts of vessels on dolphin behaviour (Peters & Stockin 2016, Peters 2018), prompted DOC, in partnership with a collective of local hapū, to establish the Pēwhairangi/Bay of Islands Marine Mammal Sanctuary (BOIMMS). The sanctuary introduced rules to control the way in which vessels can interact with marine mammals via various restrictions on vessel behaviour and the establishment of 'marine mammal safe zones' where all vessels must travel less than 5 knots (Figure 2-1). Understanding the effectiveness of the BOIMMS is critical to the management of bottlenose dolphins in the Bay of Islands.

The information on key population demographics in the Bay of Islands, including local population abundance and apparent survival, is highly outdated, with the most recent estimates being generated for 2012 (Hamilton 2013). Local abundance in 2012 was estimated at 44 (95% CI = 39–50) individuals and apparent annual survival was 0.63 (95% CI = 0.53–0.72) (Hamilton 2013). Subsequent research has provided census-based estimates (i.e., direct counts) of the number of marked individuals in the local population (Peters & Stockin 2016, Guerin 2020, Guerin 2022), that are not directly comparable to previous abundance estimates and no estimates of survival or temporary emigration have been undertaken since 2012. Thus, DOC and their iwi partners are currently lacking consistent information for the management of the Bay of Islands local population. Further, there is no information on the efficacy of the BOIMMS to aid in local population recovery.

Research to update the population demographics and spatial ecology of bottlenose dolphins in the Bay of Islands has been contracted by DOC to a research consortium involving the Far Out Ocean Research Collective (Far Out) and the National Institute of Water and Atmospheric Research Ltd. (NIWA). Specific aims of this project are:

1) To carry out field work focused on bottlenose dolphins in the Bay of Islands to provide updated information on demographic parameters, spatial ecology and compliance with the sanctuary rules.
2) To synthesize the full time-series of existing data on population demographics and spatial ecology across all institutions that have conducted research on the local population and to generate single, coherent databases of photo-identification, encounter, reproductive parameters and effort data.
3) Use the long-term databases listed above to undertake analysis to update population demographic parameters for the local population, including establishing long-term trends in population status.
4) Use the long-term databases to investigate spatial and temporal variability in distribution and habitat use.
5) Analyse data collected during field work for this study to determine level of compliance with sanctuary rules across different categories of vessel type.

The delivery of this study will enable DOC and their partners to make evidence-based decisions on the management of this important local population.

2 Methods

2.1 Fieldwork

Field surveys took place between March and June 2024 to provide updated information on the population and spatial ecology of bottlenose dolphins in the Bay of Islands and to assess compliance of vessels to the rules of the BOIMMS. The occurrence, distribution, and behaviour of dolphins within the sanctuary were assessed using a combination of vessel-based surveys and passive acoustic monitoring. The following sections provide specific details on the methods used during the field campaign.

2.1.1 Study area and BOIMMS

The Bay of Islands (35°14’S, 174°06’E) is located on the east coast of Aotearoa New Zealand’s North Island. The area covers approx. 180 km^2 (Heath 1976) and is characterised by 144 islands and islets. The Cape Brett Peninsula protrudes approx. 20 km into the South Pacific and intersects the warm East Auckland Current, providing a large catchment area for nutrients (Baker & Madon 2007). Additionally, four river systems drain into the area. Water depth within the islands is generally <20 m, gradually reaching 80 m towards the seaward boundary of the BOIMMS. The seafloor consists primarily of soft sediment, but some rocky reefs exist in central parts and fine sediment dominates the estuaries and the deeper parts of the main basin (Booth 2020). Sea surface temperature reaches 22°C in summer, dropping to 13°C in winter (Booth 1974).

The study area has long-standing whale and dolphin watching operations and is subject to a set of specific rules regarding interactions with marine mammals, including bottlenose dolphins. A permit, issued by DOC under the Marine Mammals Protection Regulations 1992, is required for commercial operators to interact with marine mammals within a 300 m radius. Vessels with permits are deemed ‘permitted’ whilst all other vessels (including recreational vessels) are deemed ‘non-permitted’. Under permit conditions, permitted vessels may interact with marine mammals but interactions before 12pm and within the specific ‘Safe Zones’ (see below), are restricted.

The Te Pēwhairangi (Bay of Islands) Marine Mammal Sanctuary was instated in 2021, accompanied by a range of area specific restrictions. All non-permitted vessels within the BOIMMS are required to come to a full stop if they find themselves within 300 m of any marine mammal and need to remain stationary until all marine mammals have moved beyond the 300 m distance. Additionally, two marine mammal safe zones (Figure 2-1) were instated. Within these zones, a speed restriction of 5 knots applies to all vessels (permitted and non-permitted). Lastly, swimming within 300 m of any marine mammal is prohibited. In addition to the BOIMMS restrictions, permits issued to commercial operators under the Marine Mammals Protection Regulations 1992 exclude the safe zones, preventing those operators from interacting with marine mammals in those areas

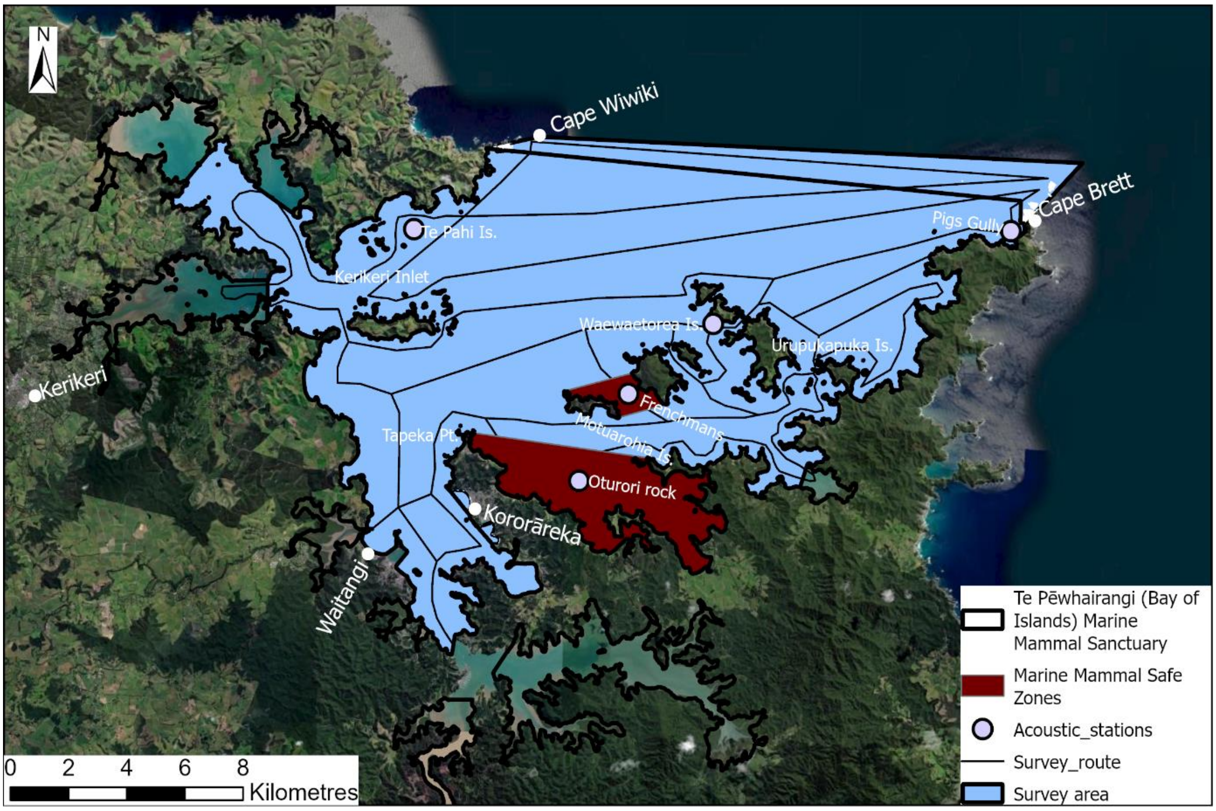

Figure 2-1: Study area and survey route. The study area within the Bay of Islands Marine Mammal Sanctuary. A generalised survey route is shown to cover the full extent of dolphin habitat within the sanctuary along with the acoustic monitoring stations where CPODs were deployed. Stations are named: Pigs Gully (near Cape Brett), Waewaetorea Is. (west of Urupukapuka Island), Frenchmans (between Motuarohia and Moturua Islands), Oturori rock (between Tapeka Pt and Manawaora Bay) and Te Pahi Is. (western BOI).

2.1.2 Vessel-based surveys

Vessel-based surveys were conducted aboard the research vessel (RV) CQRT III, a 7.6 m power driven Seamaster aluminium vessel with a single 225 hp outboard engine. Survey speed was approx. 12 knots except for the two designated 'marine mammal safe zones' where speed was dropped below the mandatory 5 knots. Surveys were conducted during Beaufort scale sea state ≤3. The crew consisted of a skipper and two experienced observers who used a continuous scanning method (e.g., Mann 1999) to detect bottlenose dolphins, looking for cues including splashes, fins, blows or bird activity. Due to the current low occupancy of bottlenose dolphins in the study area (Hartel et al. 2014, Peters & Stockin 2016, Guerin 2022), opportunistic reports from tour vessels were also used to supplement encounters with dolphin groups (as per Tezanos-Pinto 2009). This approach maximised the opportunity for encounters and ensured a sufficient sample size for population demographic analysis. In addition to the visual surveys, a directional hydrophone was used to detect dolphin vocalisations. For this, the RV was stopped with the engine turned off and a 360° directional scan was carried out using a custom-made hydrophone with a detection radius of approximately 5 km (depending on sea conditions).

Due to the convoluted shape and boundaries of the study area, systematic line-transect surveys were not deemed an appropriate use of effort (see Hartel et al. 2014). Instead, surveys followed a semi-systematic approach without a fixed route but ensuring even coverage of the study area over a two-day period. A generalisation of the survey route completed every 2 days of survey effort is shown in Figure 2-1.

2.1.3 Encounter approach

Upon encountering bottlenose dolphins, a staged observation protocol was implemented, following Guerra et al. (2014), to assess potential effects of the research vessel (RV) on dolphin behaviour. The RV maintained a distance of 300 m for the first 20 minutes, recording behavioural states every five minutes (see below), while remaining stationary or travelling at idle speed only to maintain visual contact with the dolphins. After the initial 20-minute distant observation period, the dolphins were approached within 50 meters. At this point, data collection began, including the recording of date, time, geographic coordinates, and group-specific information such as group size, presence of calves, and with a continuation of logging behavioural observations every 5 minutes. Photo-identification of individual dolphins (photo-ID) procedures were also initiated.

2.1.4 Recording dolphin behaviour

Behavioural information was collected using focal follow techniques previously used to study the behavioural ecology of the local population (Constantine et al. 2004, Peters & Stockin 2016, Peters 2018, Guerin 2022). The behavioural state of dolphin groups was determined by conducting a two-minute scan of the entire group and determining the overall behavioural state of the majority (>50%) of the group. The following six behavioural states were defined as per Table 2-1. Behavioural states were recorded at five-minute intervals during an encounter.

Table 2-1: Behavioural states. Definition of behavioural states used to classify bottlenose dolphin behaviour during focal follows.

Behavioural state	Definition
Diving/foraging (DIV)	Direction of movement varies; individuals dive synchronously for long intervals and perform steep dives (arching their backs at the surface); group spacing varies. OR surface foraging behaviour is evident (corral or chase prey).
Milling (MIL)	Little or no net movement; low coordination in orientation (individuals surface facing different directions); short dive intervals; individuals relatively loosely grouped
Resting (REST)	Group moves slowly in a constant direction (≤2 knots), swimming with short, relatively constant, synchronous dive intervals; individuals tightly grouped
Socialising (SOC)	Diverse interactive behavioural events observed, such as sexual, inter-individual inter - actions and aerial behaviours; dive intervals vary; group spacing varies
Travelling (TRA)	Group moves steadily in a constant direction (>2 knots), swimming with short, relatively constant dive intervals; group spacing varies
Interacting with boat (INTR)	Group closely approaches boat; dolphin−boat interactive behavioural events observed, such as bow-riding and stern wake-riding

2.1.5 Vessel compliance and interactions

The presence, number, type and speed of all vessels within 300 m of dolphin groups were recorded. We define an interaction as the presence of a vessel within 300m of a dolphin group. Vessels were placed into one of the following categories: Permitted power vessel, permitted sailing vessel, non-permitted recreational power vessel, non-permitted recreational sailing vessel. The relative proximity of vessels to dolphin groups was estimated using a laser range finder, and binoculars with inbuilt compass and reticules. Approximate vessel speed was estimated by taking multiple fixes of individual vessels, obtaining bearings from the binoculars and distance from the rangefinder. Multiple fixes (i.e., bearings/distances) were obtained for each vessel and the mean distance between fixes was calculated which was divided by time to derive an approximate speed. In addition to behavioural categories recorded during the staged encounter approach, all interactions between dolphins and vessels (such as bow-riding) were recorded to assess their relative impact on the behaviour of the dolphins.

2.1.6 Monitoring vessel compliance within the marine mammal safe zones

In addition to assessing vessel compliance with respect to direct interactions with bottlenose dolphins, compliance with the 5 knot speed restrictions within the two marine mammal safe zones was assessed. The RV was stationed centrally within each zone for a 20-minute period and the speed and type (e.g., permitted, power driven etc.) of every vessel detected were recorded as either *compliant* or *non-compliant* (according to the estimated vessel speed, see previous section). At least one zone was assessed per survey day.

All data were entered into a purpose-built tablet-based app using Cybertracker software (Cybertracker Ltd.).

2.1.7 Photo-identification

Standard photo-ID methods were applied (e.g., Würsig and Jefferson 1990) using digital SLR cameras with 100-400 mm zoom lenses to identify individual bottlenose dolphins. Notches on the trailing and/or leading edges of the dorsal fin were used as primary identification features, with scratches and/or scars used as secondary identification features. A capture was defined as one or more useable photographs obtained during a discrete encounter. Photo-ID efforts continued until observers were confident that each member of the group had been photographed, including those with no obvious marks (see calculation of mark ratio in section 2.4.1 below). Photographs were graded following Zaeschmar et al. (2020), according to four categories depending on their quality as follow: 1 = *excellent*, 2 = *good,* 3 = *fair*, and 4 = *poor*. Only images of quality ratings 1 and 2 were used in all analyses. Photographic quality was based on focus, size of the dorsal fin relative to the frame, exposure (contrast between dorsal fin and background), interference (i.e. water splashes) and relative angle to the dolphin. Additionally, individuals were graded according to the distinctiveness of their dorsal fins, ranging from 1 to 4 (*very distinctive* to *not distinctive*). Successful identifications were then matched to previously identified individuals and all 'new' (previously unidentified) individuals were assigned a catalogue number and entered into the photo-ID catalogue (see Tezanos-Pinto et al. 2013 for more details). Individuals were then matched to existing catalogues of the study area and adjacent areas to determine ranges, habitat use and site-fidelity (see Section 2.2 Integrating Datasets below).

Particular attention was given to the documentation of calves, with individuals showing close associations (<1 body length) or swimming in the echelon position being captured photographically to ascertain the identity of any accompanying dolphin. Following Tezanos-Pinto et al. (2015), mother identity was confirmed for each calf by observing mother – calf associations over at least two subsequent and independent encounters, occurring over different days.

Associations among individuals were assessed using Netdraw 2.177 (Borgatti 2002). A sociogram was produced, applying a spring-embedded layout which places more connected individuals at the centre of the network and those with fewer connections at the periphery. This analysis was conducted for data collected from March to June 2024 only.

2.1.8 Acoustic monitoring

Five echolocation data loggers (CPODs, Chelonia Ltd.) were deployed throughout the study area to assess the presence and behaviour of dolphins (from March to June 2024). Vessel presence was also assessed by logging the pulses from vessel-based echosounders, which can provide a relative measure of vessel density. Devices were positioned widely across the study area to ensure even and representative coverage and in areas with varying vessel restrictions. One device was deployed in each of the two marine mammal safe zones to assess dolphin use of these areas.

CPODs were deployed in each location at the onset of the fieldwork programme (March 2024), suspended approximately 4 m off the seafloor on fixed moorings marked with a small surface float. CPODs were retrieved and downloaded at the end of the fieldwork programme at the end of May 2024. All acoustic data was downloaded from the customised SD cards in the software CPOD.exe (Chelonia) for analysis.

2.2 Integrating datasets

Bottlenose dolphin research in the study area was initiated in 1994 and various datasets have been produced in the meantime. The present study seeks to integrate all available datasets on population demographics, reproductive data and spatial distribution into a single, unified resource to enable a more efficient and accurate assessment of long-term trends within the local population. The three types of information gathered across multiple institutions includes 1) photo-identification data on the date of sightings of individual dolphins with identifying features, 2) encounter data on sightings of groups of bottlenose dolphins, 3) reproductive data on the fate of confirmed calves born into the local population.

A master encounter dataset was produced to include the full information on encounters with bottlenose dolphins from historical research in the Bay of Islands. The master encounter dataset includes the date and time of each encounter, the encounter locations, spatial coordinates (where available), group size information (mean group size, minimum and maximum where available), the presence of neonates and calves and any key behaviours observed during an encounter. When reviewing the historical datasets, the first entry for a given dolphin group was used for the master encounter dataset (e.g., multiple observations of the same group were not considered from historical datasets that used focal follow techniques to investigate dolphin behaviour).

A master reproductive (i.e. calf mortality) database was configured following a template provided by G. Tezanos-Pinto, that was used in a previous assessment of calf mortality for this local population (Tezanos-Pinto et al. 2015). The database consists of entries of confirmed calves (i.e., with known mother identity observed over two or more independent encounters), the date (or year) of birth, and whether the calf survived to one- or two-years post birth. Data were integrated into this template from raw records of mother–calf associations held by the University of Auckland between 2006 and 2012, by Massey University between 2012 and 2015 and by TriOceans Ltd. between 2015 and 2022. Calf entries from fieldwork undertaken during this study were also added. All records of calves where the mother identity could not be confirmed were discarded. However, calves with unknown fate (i.e., those where the mother was not seen within 1 or 2 years post-birth) were retained in the database as these entries can be used for estimating reproductive rate (but not calf mortality; see section 2.4.1).

A master effort database was also generated integrating daily effort, recorded as hours on effort (the most consistently used metric) for each of the 1605 survey days in the Bay of Islands between 1994 and 2024.

2.2.1 Master photo-identification database

All individuals from the existing photo-identification catalogues for the Bay of Islands were matched to produce a single master catalogue. Matching was aided by finFindR software (Thompson et al. 2022), through R version 4.0 (R Core Team 2023). We used finFindR to rank the 15 most likely individuals from the selected catalogues, which were then manually checked to confirm a match. If a high-certainty match was not provided by finFindR, the individual was matched manually against all catalogues before being entered as a 'new individual' (i.e., not previously encountered in the Bay of Islands). All previously catalogued individuals retained their original catalogue numbers but were also assigned a new catalogue number within the new master-catalogue.

The format of sighting records for bottlenose dolphins varied significantly among historical researchers. Thus, in this study we unified the master photo-ID (e.g., sightings history) dataset across all individuals using the new ID catalogue number that had been cross-matched with all other catalogues. Available datasets from the University of Auckland (1994–2012) held specific information on the date, time and ancillary encounter information (e.g., calf associations) for each ID. Datasets from Massey University and TriOceans Ltd. were provided in the form of approx. 28,000 matched photo-ID photographs with the ID of individuals listed in the filename of each image. The date of each photograph was similarly listed in the filename as well as being listed in the metadata of each image. The two dates were cross-checked and found to be largely (<3% difference) in agreement. Any disparities between dates were corrected by determining the likely date of photo-ID given the database of recorded daily effort. Any likelihood of incorrect date (typically 1 day of difference) impacting analyses was carefully checked by comparing photo-ID data formatting for the robust design (see below) using both date formats and was found to be negligible.

The master sightings history contains columns for every day of photo-identification research in the Bay of Islands between 1993 and 2024, with rows denoting the ID (by new catalogue number) of each individual.

2.3 Photo-Identification data curation

Images of insufficient quality and/or individuals lacking distinct distinguishing features were excluded from the dataset (i.e. that were of photo quality >2, distinctiveness >2 refer to section 2.1.7). Additionally, any individuals known or suspected to belong to the oceanic ecotype were omitted. from the analyses.

While the oceanic ecotype is known to occur in the study area, particularly during the warmer months, its occurrence is primarily restricted to deeper waters. Previous research shows that interactions between the coastal and the oceanic ecotypes are limited, and that the two should be considered as distinct management units (Zaeschmar et al. 2020). The two ecotypes can be distinguishable by (in the oceanic ecotype) the presence of cookie cutter shark (*Isistius* spp.) bite marks, darker colouration and a larger, more robust body. To ensure the integrity of the analysis, all individuals that were only identified in encounters involving known or probable oceanic bottlenose dolphins were excluded. These criteria guarantee that the final photo-identification database includes only individuals of sufficient distinctiveness and image quality to enable reliable re-identification over long time periods and that are highly likely to belong to the coastal ecotype frequenting the study area.

2.4 Analysis

2.4.1 Population demographics

Population demographic parameters were estimated using two approaches:

1. A 'snapshot' of local population abundance was produced based solely on data from the most recent field programme, providing the most up-to-date estimate of current local abundance.

2. The full photo-identification database, spanning a subset of the entire study period from 1997 to 2024, was used to model a range of demographic parameters, allowing for an analysis of variability in population dynamics over time.

Population abundance ‘snapshot’

For the local population abundance ‘snapshot’, we used data gathered during the field surveys undertaken from March to May 2024. The ‘snapshot’ was calculated using a comparatively small window for data collection and thus does not reflect the true number of individuals in the local population, and was used to provide a minimum estimate of the number of individuals using the study area during the fieldwork undertaken by this study (i.e., between March and May 2024 inclusive). We estimated local population abundance using a mark-recapture (MR) ‘closed’ model. Given the relatively short duration of sampling, we split encounters into two groups: an initial ‘mark’ period from the 23rd of March until 17th of April, and a recapture period from the 23rd of April until the 23rd of May. The cutoff between periods was set in order to apportion approximately one month to each period, standardise effort, and to allow a period of re-mixing among individuals (there were five days of no surveys between periods). Encounter-specific capture histories for the 24 individually marked dolphins encountered during our surveys were merged to create a sighting matrix for each ID denoting its presence or absence in the mark or recapture period.

Closed MR models were fit in RMark (Laake 2013) within R. A closed formulation is most appropriate for the snapshot demographic assessment as: 1) there are limited data with which to fit more complex models; 2) assumptions around population closure (birth/death and emigration/immigration) are likely upheld given the short study period; 3) the method aligns with the model formulation for the full dataset (i.e., Robust design) that treats within-primary-period population dynamics as closed; and 4) closed MR models are routinely used to monitor population abundance in other NZ bottlenose dolphin populations with varying residence patterns (Bennington et al. 2020). The closed MR model used here is a modification of the original Lincoln-Peterson estimator of closed populations (Otis et al. 1978). The core parameters of the closed MR model formulated in RMark are:

p – Probability of first capture

c – Probability of recapture

$f0$ – number of unsampled individuals

Typically, in MR modelling a range of parameter specifications are used to test candidate model designs to derive the best fit to the capture history data (White & Cooch 2001). In closed MR formulation, *p* and *f0* are required to be constrained (e.g., constant) in order to be able to accurately derive N̂ (population abundance) (Lukacs 2009). In addition, the parameter *c* could not be time varying due to the single recapture period. Thus, a single model formulation with each of the key parameters constrained was fit in R.

Local population abundance (N̂) is derived from the parameter estimates of the closed model,

$$\hat{N} = f0 + M_{t+1}$$

where M_{t+1} is the total number of marked individuals across the mark and recapture period(s).

Robust design – long-term population modelling

Long-term variability in population demographics of bottlenose dolphins in the Bay of Islands was investigated using the robust design (RD) mark-recapture framework (Pollock 1982), implemented in RMark (Laake 2013). The robust design allows for the incorporation of the oftentimes confounding effects of temporary emigration upon the estimation of survival rates and ultimately, abundance, making it an ideal framework for assessing demographics in the Bay of Islands with known variability in the rates of occurrence among individual dolphins (Constantine & Baker 1997). The robust design separates capture histories into primary and secondary periods. Within primary periods, the population is assumed to be closed to births, death and immigration/emigration. The population is assumed to be open between primary periods, allowing for estimation of dynamic changes in population demographic rates. At least two secondary periods are nested within each primary period, with these periods representing individual closed samples of the population.

While photo-identification has been used throughout the 30+ years of research in the Bay of Islands, not all photo-ID effort is suited for analysis using the robust design. To provide estimation of demographic parameters at sufficiently broad temporal scales to allow meaningful inference, the primary periods for analysis in this study represented the austral seasons: summer (Dec–Feb), autumn (Mar–May), winter (June–August) and spring (September–November). Seasonal or annual primary periods are the most frequently used in RD analysis (Silva et al. 2009; Smith et al. 2013; Fruet et al. 2015; Santostasi et al. 2016; Methion & López 2018; Durden et al. 2021). Secondary periods are required to be instantaneous (with respect to the length of the study) snapshots of the population (Kendall 2006), yet also require sufficient effort to ensure dolphins were photographed if present. We developed decision rules around the amounts of days and effort that could be represented in a single secondary period:

1. Individual effort days we required to have documented survey effort with at least 1 hour of photo-ID effort to be considered as part of a secondary period;
2. Secondary periods are at least 2 and no longer than 10 days in length, with a secondary period requiring at least 2 full days of survey effort. The full Bay of Islands area can be surveyed in a minimum of 2 days of effort; and
3. Effort days are largely continuous (i.e., with limited gaps) within secondary periods.

Seasonal primary periods that had at least two sequences of photo-ID effort that meet the requirements for secondary periods were subsampled from the long-term photo-identification dataset (hereafter RD subset). The time between consecutive primary periods was variable (median =

29), with a minimum of 7 days to ensure some mixing of individuals between periods. Forty-seven primary periods from 1997 to 2024 contained sufficient photo-ID effort for inclusion in the robust design analysis. The number of secondary periods within each primary period varied between 2 and 7 (Table 2-2).

Table 2-2: Summary of seasonal primary periods for analysis using the robust design.

Primary period	Date range	n secondary periods	n days	Effort (hrs)	n dolphins identified
Summer 1997	05 Dec 1996 - 28 Feb 1997	4	18	104.4	94
Autumn 1997	27 Mar 1997 - 25 May 1997	3	9	40.3	79
Winter 1997	07 Jun 1997 - 31 Aug 1997	2	5	22.4	57
Spring 1997	03 Oct 1997 - 03 Nov 1997	4	14	62.2	87
Summer 1998	21 Dec 1997 - 26 Feb 1998	5	19	98.8	84
Autumn 1998	04 Mar 1998 - 14 Apr 1998	5	20	94.4	128
Winter 1998	21 Jun 1998 - 23 Aug 1998	5	19	80.3	55
Spring 1998	11 Sep 1998 - 22 Nov 1998	4	13	38.8	43
Summer 1999	01 Dec 1998 - 05 Feb 1999	5	25	119.5	61
Autumn 1999	08 Mar 1999 - 12 May 1999	5	22	82.5	77
Winter 1999	01 Jun 1999 - 26 Aug 1999	5	23	95.7	75
Spring 1999	11 Oct 1999 - 25 Nov 1999	3	10	35.9	35
Summer 2000	09 Dec 1999 - 29 Feb 2000	5	22	54.3	41
Winter 2003	03 Jun 2003 - 21 Jul 2003	3	10	52.9	29
Spring 2003	25 Sep 2003 - 13 Nov 2003	3	8	38.4	22
Summer 2004	01 Dec 2003 - 14 Feb 2004	3	9	41.5	65
Winter 2004	06 Jun 2004 - 31 Aug 2004	3	11	73.2	20
Spring 2004	21 Sep 2004 - 22 Nov 2004	5	27	210.4	96

Primary period	Date range	n secondary periods	n days	Effort (hrs)	n dolphins identified
Summer 2005	08 Dec 2004 - 26 Feb 2005	3	12	89.2	69
Winter 2005	10 Jun 2005 - 28 Jul 2005	2	8	49.8	17
Spring 2005	13 Sep 2005 - 20 Nov 2005	5	29	181.4	67
Autumn 2006	21 Mar 2006 - 01 May 2006	2	8	46.1	50
Winter 2009	18 Jun 2009 - 27 Aug 2009	5	28	161.8	83
Spring 2009	15 Sep 2009 - 25 Nov 2009	4	18	116.1	86
Summer 2010	03 Dec 2009 - 07 Jan 2010	3	15	98.4	99
Summer 2013	08 Dec 2012 - 23 Feb 2013	5	41	411.2	77
Autumn 2013	01 Mar 2013 - 30 May 2013	7	58	443.2	85
Winter 2013	07 Jun 2013 - 29 Aug 2013	7	63	327.4	79
Spring 2013	04 Sep 2013 - 30 Nov 2013	7	63	324.2	99
Summer 2014	15 Dec 2013 - 27 Feb 2014	5	49	275.4	89
Autumn 2014	01 Mar 2014 - 31 May 2014	7	70	332.6	93
Winter 2014	03 Jun 2014 - 27 Aug 2014	7	63	256.0	74
Spring 2014	01 Sep 2014 - 30 Nov 2014	7	70	307.1	85
Summer 2015	07 Dec 2014 - 28 Feb 2015	7	61	258.2	72
Autumn 2015	03 Mar 2015 - 20 Apr 2015	4	35	143.5	37
Spring 2017	13 Oct 2017 - 26 Nov 2017	3	13	28.7	18
Summer 2018	03 Dec 2017 - 27 Feb 2018	4	22	62.6	24
Summer 2019	27 Dec 2018 - 05 Feb 2019	3	10	12.2	23
Winter 2019	16 May 2019 - 07 Aug 2019	3	9	23.0	16

Primary period	Date range	n secondary periods	n days	Effort (hrs)	n dolphins identified
Spring 2019	27 Oct 2019 - 29 Nov 2019	4	16	20.3	21
Summer 2020	04 Dec 2019 - 19 Feb 2020	5	19	40.3	17
Summer 2021	04 Dec 2020 - 25 Feb 2021	3	12	15.5	47
Spring 2021	22 Mar 2021 - 14 May 2021	2	7	31.0	14
Autumn 2021	07 Nov 2021 - 14 May 2021	3	13	21.9	15
Autumn 2022	15 Apr 2022 - 14 May 2021	2	6	18.4	14
Winter 2022	07 Jun 2022 - 15 May 2021	2	7	12.8	9
Autumn 2024	25 Mar 2024 - 17 May 2021	5	20	157.5	26

The core parameters of the robust design (Kendall 2006) are:

φ= *Survival.* Probability that an individual survives between periods;

p - Probability of first capture. The probability of being detected given an animal was not previously detected within a given sampling period;

c – Probability of recapture. The probability of being detected given an animal was previously detected within a given sampling period;

γ' and γ'' = *Temporary emigration*. The probability of being outside the study area during the primary period given that the animal was not present (γ') or was present (γ'') during the previous primary period (t-1). γ' can be considered the probability of remaining outside the study area, while γ'' is the probability an animal moves outside of the study area;

f0 – Number of unsampled individuals.

A range of parameterisations of these core parameters can be considered in order to determine the best fit of RD models to the observed capture histories and to better uphold the assumptions of the open and closed components of the model framework. These assumptions include equal probability capture *(p)* and recapture (*c*) among individuals (i.e., no capture heterogeneity), the closure of the population within primary periods, minimal rates of transience and a lack of behavioural responses to capture/recapture (e.g., 'trap happy/shy' responses) (Kendall & Pollock 1992; Kendall et al. 1997; Kendall 2006). The RD has no specific checks for the upholding of these assumptions or goodness of fit. Thus, to check these assumptions, we transformed our RD subset of capture histories to represent the open components of the model (i.e., pooled data into primary periods) and the closed components of the model (i.e., pooled data into separate, secondary periods). Population closure within primary periods was investigated using the function *closure.test* within R package *secr* (Efford 2024) which confirmed closure all but 8 primary periods (Appendix A). Heterogeneity in capture probabilities was assessed using the functions Test 2.CL and Test 2.CT and transience was assessed with Test3.SR in R package R2Ucare (Gimenez 2022). Tests indicated the presence of significant heterogeneity in capture probabilities and transience (p values <0.05 for both tests) (see Appendix A).

To account for variation in capture probabilities among individuals, we implemented a modified version of the classic robust design framework that accounts for such variation by including a finite mixture model within the estimation of *p* and *c* (Kendall 2006). In RMark, this framework was implemented using the "RDHet" specification that includes two latent classes within the estimation of capture probabilities, with each class having higher or lower probability of capture (Pledger & Phillpot 2008; Gimenez et al. 2018). The proportion of individuals within each latent class is included as an additional parameter (π) that is estimated during the model fitting process.

Transience confounds survival by exerting downward bias on parameter estimation as transience (i.e., being recorded only once) is interpreted by the models as mortality (Kendall 2006; Silva et al. 2009). We accounted for transience by the inclusion of a covariate (*transience*) in the estimation of survival that reflects the time since first capture for each individual (Silva et al. 2009; Durden et al. 2021). The '*transience'* covariate enables the estimation of survival separately for the first primary period that individuals are encountered, which includes all individuals encountered only once in the study period, separating survival estimates for 'resident' dolphins from more transient individuals (Silva et al. 2009; Durden et al. 2021).

We developed a full suite of candidate models to determine the best performing combinations of parameters and to enable an objective assessment of a range of parameterisations upon our key demographic parameters of interest: survival, abundance and temporary emigration. Parameterisation included:

- Survival (ϕ) to time varying – ϕTr(t) or constant – ϕTr(.), where Tr represents our transience covariate.
- Capture probability (*p)* to time varying – pMix(t), constant – pMix(.) or including a covariate 'effort' to represent the amount of survey effort (in hours) per primary period (pMix:effort). 'Mix' represented the mixture model covariate.
- Mixture probability (π) was configured as either time varying – π(t) or constant π(.),
- The number of unsampled individuals (f0) was either time varying - f0(t) or constant - f0(.).
- We included two forms of temporary emigration for assessment: random temporary emigration (γ″ = γ′), where emigration does not depend on the previous state of an individual and Markovian temporary emigration (γ′ ≠ γ″) where rates of temporary emigration does depend on the previous state of an individual (i.e., whether it was within or outside of the study area during the previous capture period) (Kendall 2006).
- Recapture probability (*c*) was set to be equal to capture probability (*p)* because there was no behavioural response to recaptures (Kendall 2006).

All time varying versions of the parameters were configured to vary according to primary periods as the incorporation of heterogeneity in capture probabilities does not enable determination of *p* according to secondary periods. The exception to this was the temporary emigration parameters where time varying specification caused the parameter estimates for several periods to be non-identifiable. Thus, a categorical variable denoting emigration rates by decadal periods was generated, with an early (1994 – 2003), mid (2004 – 2013) and late (2014 – 2024) time period configured for both γ' and γ'' to test the hypothesis of variability in emigration rates over the study period.

The full suite of 49 candidate models were fit in RMark and evaluated using the Akaike Information Criterion for small sample sizes and adjusted for overdispersion (QAICc) (see below). A model selection table including QAICc, ΔQAICc, model weight and negative log likelihood was generated to determine the most parsimonious model given our data. 'Real' parameter estimates (i.e., parameters included within the model specifications), and their associated variance were extracted from the most parsimonious model. The abundance of marked individuals ($\hat{N}$) for each primary period and its associated variance was extracted from the most parsimonious model as a derived parameter, and was scaled by the mark ratio (see below) to generate an estimate of total local population abundance (i.e., for marked and unmarked individuals) for each primary period.

Goodness of fit for the robust design analysis was assessed using the $\hat{C}$ statistic. $\hat{C}$ was calculated via parametric bootstrapping of simulated capture histories based on the parameter estimates of a Cormack-Jolly-Seber open population model fit in RMark. Parametric bootstrapping was carried out 1000 times and median $\hat{C}$ calculated. Median $\hat{C} > 1$ indicates overdispersion of the binomial capture histories may impact model fit. In such cases, model fitting in RMark can be adjusted by specifying the 'chat' argument and undertaking subsequent model selection using the Quasi-likelihood Akaike Information Criterion (QAIC) (Anderson & Burnham 2002).

Calculation of mark ratio

Mark ratio ($\hat{P}_m$) and associated variance were calculated based on the proportion of photographs of marked individuals out of the total number of high-quality photographs taken in the field following protocols established by Gormley et al. (2005).

$$\hat{P}_m = \frac{\sum_{i=1}^{k} \frac{I_i}{T_i}}{k}, var(\hat{P}_m) = \left(\sum_{i=1}^{k} \frac{\hat{P}_{m_i}(1 - \hat{P}_{m_i})}{T_i} \right) \Big/ k^2$$

where I_i is the number of photographs of marked dolphins, T_i is the total number of good quality photographs taken during the ith sampling day and k is the total number of sample days for which I/T was calculated. Mark ratio was used to scale the local population abundance estimates to represent the number of both marked and unmarked individuals in the local population at the time of our survey and within the constraints of the available data (Gormley et al. 2005; Tezanos-Pinto et al. 2013).

The abundance of marked dolphins ($\hat{N}_m$) is then scaled by the mark ratio ($\hat{P}_m$) to obtain the total abundance ($\hat{N}_{total}$):

$$\hat{N}_{\mathrm{total}} = \frac{\hat{N}_{\mathrm{m}}}{\hat{P}_{\mathrm{m}}}$$

with variance and standard error (SE) as:

$$var(\hat{N}_{total}) = (\hat{N}_{total})^2 \left(var\ (\hat{N}_m)/(\hat{N}_m)^2 + var\ (\hat{P}_m)/(\hat{P}_m)^2 \right)$$

$$SE(\hat{N}_{total}) = \sqrt{var(\hat{N}_{total})}$$

Log-normal confidence $\hat{N}$ lower and upper bounds (Burnham 1987) were calculated using:

$$\hat{N}_{lower} = \frac{\hat{N}_{total}}{C}, \hat{N}_{upper} = \hat{N}_{total} \cdot C, \text{where } C = exp\left(z_{\alpha/2}\sqrt{log_e\left[1 + \left(cv(\hat{N}_{total})\right)^2\right]}\right)$$

where $z_{\alpha/2}$ is the normal deviate, α = 0.05 ($z_{\alpha/2} \approx 1.96$ and cv is the coefficient of variation $\hat{N}_{total}/\sqrt{var(\hat{N}_{total})}$

Mark ratio has been calculated multiple times for the local population using the procedure outlined above; by Tezanos-Pinto et al. (2013) for the period 2003 to 2006, by Hamilton (2013) for 2007 to 2010 and using data collected by this study in 2024. Mark ratio can vary according to a range of factors including the age demographics of the local population and the exposure to human activities that result in marks (e.g., commercial fishing). To ensure the most accurate estimates of local population abundance (of marked and unmarked individuals), for each primary period, we scaled each abundance estimate by the mark ratio most closely associated with that period (by year). In other words, abundance estimates for primary periods between 1997 and 2006 were scaled by 0.72 (Tezanos-Pinto et al. 2013), between 2007 and 2017 by 0.88 (Hamilton 2013), and between 2018 and 2024 by the MR calculated by this study.

Calf mortality

Calf mortality was estimated following Tezanos--Pinto et al. (2015) using data on the known fate of calves with confirmed mother identity, over two consecutive and independent sightings. We pooled data directly from Tezanos-Pinto et al. (2015) and added the data on the fate of confirmed calves post 2006 (the end date of that study) to establish our master reproductive database. For all entries, close association of a neonate, calf or juvenile with an adult dolphin observed 2 or more subsequent encounters over at least 2 days was used to confirm mother identity. Birth date was assigned as the date of first sighting for neonates, which is representative as a 'minimum' age. The analysis considered only neonates or very young calves that could have the year of birth established with certainty, via confirmation over two or more consecutive sightings. A calf was deemed to have died if its mother was resighted without its calf and the calf was less than 1.5 years old at the time of the mother's resighting. The minimum age of weaning in bottlenose dolphins is approximately 18 months (Wells & Scott 1990; Smolker et al. 1992).

Only calves with known fate were used to calculate mortality rates. As such, calves with unknown fates (i.e., those with mothers that had not been resighted before possible weaning) were removed from the analysis. Apparent first year calf mortality was calculated as the proportion of young of the year that were assumed to have died of the total number of young of the year, for a given year. Apparent second year calf mortality was calculated as the proportion of calves likely to have died before reaching 2 years of age, for a given year. Calf mortality rates were calculated for each year over which data on mother and calf association had been reliably recorded following two or more consecutive and independent sightings. Years are represented by calving season (i.e., July to June) rather than calendar year to avoid splitting the summer (when most calves are born) across years (i.e. '2001' runs from 1st July 2001 to 30th June 2002). Wilson's modification of binomial variance estimates for small sample sizes (Wilson 1927), were used to calculate 95% confidence intervals around the annual calf mortality rates.

For calves sighted during the field programme for this study, we determined mother identity following the protocols detailed above. Young of the year and very young calves were assigned to birth dates and information on confirmed associations of calves and juveniles was used to confirm mortality rates of calves recorded to have been born in previous years' surveys. Note that there is a gap between the last surveys on this local population (winter 2022) and the present study, thus first year calf survival cannot be calculated for 2022.

Reproductive rate

The reproductive rate, or birth rate, was calculated based on prior research on this local population (Tezanos-Pinto et al. 2015) for each year with reliable data on mother–calf associations. Specifically, confirmed mother-calf association needed to occur over two or more confirmed consecutive and independent sightings. Reproductive rate was defined as the number of calves born into the local population for each reproductive season (i.e., July – June), (split at June) divided by the total number of reproductive females sighted during that year. Only calves with confirmed mother identity (i.e., seen more than twice on independent sightings with the same female) were used for the calculation of reproductive rate. The number of reproductive females seen each year was the total sum of individuals that have ever been recorded as having given birth to a calf, before the given year (sighted on more than two independent occasions with a calf). These data were provided directly from studies by Tezanos-Pinto et al. (2015) and Peters & Stockin (2016). Data between these two studies and from Guerin 2022 were generated by reviewing the number of reproductively active in the master photo identification database for all entries across a given year, while adding any females first confirmed as reproductively active post 2016.

2.4.2 Species distribution modelling

Dynamic species distribution modelling (SDM) was used to resolve the spatial and temporal distribution and to investigate the drivers of habitat use of bottlenose dolphins in the Bay of Islands. Data on the occurrence (i.e., presence) of groups of bottlenose dolphins sighted during surveys between 1994 and 2024 were pooled with randomly generated background (i.e., absence) data to model the relationship between occurrence and environmental variables and predict spatiotemporal distribution throughout the study area. A total of 762 occurrence records were available with reliable spatial and temporal information (i.e., geographic coordinates, date/time) for modelling.

While the amount of effort during each survey day in the Bay of Islands is available (Table 2-2), data on the daily distribution of effort (i.e., survey tracks) is available for only a subset of days, preventing the generation of absence points within areas known to be surveyed each day (e.g., Stephenson et al. 2023). Thus, to generate a representative sample of background data we randomly generated background points throughout study area following standard procedures (Elith & Leathwick 2009; Barbet-Massin et al. 2012) at the daily scale. A database with a total of 20,000 background points was created, with the number of points per survey day being calculated as a proportion of the total effort carried out during a given day (Derville et al. 2018). To account for biases associated with uneven distribution of survey effort (e.g., Figure 3-2), we calculated a fixed gaussian kernel around the occurrence data points for each survey day, and extracted values from this kernel to weight the contribution of background data points to model tuning (Elith et al. 2010; Fourcade et al. 2014; Stolar & Nielsen 2015). Under this approach, background data points closer to presences have a higher weight, ensuring both presences and background data have the same spatial biases (Phillips et al. 2009). This approach is often used to negate the influences of uneven sampling distribution in the development of SDMs (Elith et al. 2010; Fourcade et al. 2014; Finucci et al. 2021).

A broad range of environmental variables were pooled for the development of the SDMs (Table 2-3). The variables consisted of six static (i.e., not time-varying) variables (e.g., bathymetry and slope – see Table 2-3) that were sourced from NIWA's national environmental data repository (Stephenson et al. 2022). Four dynamic (i.e., time varying) variables were sourced from satellite remote sensing data at weekly temporal resolution. The Copernicus Climate Data Store was used to provide dynamic variables for dates prior to July 2002, and NIWA's SCENZ repository (Pinkerton et al. 2022) from July 2002 onwards (SCENZ data is sourced from the MODIS-Aqua mission and is thus not available for dates before July 2002). The spatial resolution of the dynamic variables was accessed at 4 km x 4 km and 500 m x 500 m resolution for the Copernicus and SCENZ data respectively, with these data being resampled to 250 m x 250 m to match the static variables (Table 2-3).

To ensure comparability of the two dynamic datasets, a weekly average layer was selected randomly for each of the six months within each time period used to predict dynamic distribution (see below). Using extracted cell values for the full study area, the variables were compared using a scatterplot and fitting a linear regression. The equation from this regression was used to transform the Copernicus data such that it was equivalent to the SCENZ data.

Table 2-3: Environmental variables. The environmental variables used for species distribution modelling of bottlenose dolphins in the Bay of Islands.

Variable	Name	Description	Temporal resolution	Reference
Bathy	Bathymetry	Depth of the seafloor	Static	National scale dataset; NIWA unpublished, updated in 2020
BBP	Particulate backscatter	The particulate backscatter coefficient at 555 nm (m^{-1}), which is highly correlated with turbidity measurements by optical backscatter sensors.	Weekly and monthly	Copernicus Climate Change Service 2019 for data prior to July 2002; NIWA-SCENZ, Pinkerton et al. 2022 for July 2022 onwards
BedDist	Benthic sediment disturbance	One-year mean value of friction velocity derived from estimates of surface wave statistics, median grain size and water depth. Benthic sediment disturbance from wave action was assumed to be zero where depth ≥ 200m.	Static	National scale dataset; NIWA unpublished, updated in 2020

Variable	Name	Description	Temporal resolution	Reference
BPI_fine	Bathymetric position index (fine-scale)	Bathymetric position index (BPI) is a measure of where a referenced location is relative to the locations surrounding it. Terrain metrics were calculated using an inner annulus of 12 km and a radius of 62 km.	Static	National scale dataset; NIWA unpublished, updated in 2020
CHL	Chlorophyll-a concentration	A proxy for the biomass of phytoplankton present in the surface ocean (to ~30 m depth)	Weekly and monthly	Copernicus Climate Change Service 2019 for data prior to July 2002; NIWA-SCENZ, Pinkerton et al. 2022 for July 2022 onwards
MLD	Mixed layer depth	The depth that separates the homogenised mixed water above from the denser stratified water below	Static	National scale dataset; NIWA unpublished, updated in 2020
Slope	Slope	Bathymetric slope was calculated from water depth and is the degree change from one depth value to the next	Static	National scale dataset; NIWA unpublished, updated in 2020
SST	Sea surface temperature	Blended from OI-SST (Reynolds et al. 2002) ocean product and MODIS-Aqua SST coastal product. Long term (2002 – 2021) average values at 250 m resolution	Weekly and monthly	Copernicus Climate Change Service 2019 for data prior to July 2002; NIWA-SCENZ, Pinkerton et al. 2022 for July 2022 onwards
SSTGrad	Sea surface temperature gradient	Smoothed magnitude of the spatial gradient of annual mean SST. This indicates locations in which frontal mixing of different water bodies is occurring (Leathwick et al. 2006).	Weekly and monthly	Calculated for this study, using Copernicus Climate Change Service 2019 for data prior to July 2002; NIWA-SCENZ,

Variable	Name	Description	Temporal resolution	Reference
				Pinkerton et al. 2022 for July 2022 onwards
TC	Tidal Current speed	Maximum depth-averaged (New Zealand bathymetry) flows from tidal currents calculated from a tidal model for New Zealand waters (Walters et al. 2001)	Static	National scale dataset; NIWA unpublished, updated in 2020

Presence and background data were overlaid on a 250 m x 250 m spatial grid in an Albers equal area projection. The data were matched with the location's corresponding environmental characteristics according to the spatial coordinates of each data point and date stamp, with points being matched to the corresponding week for the dynamic variables. A factor variable for 'Month' (12 levels) was also attributed to each presence and background data point. Collinearity among environmental variables was evaluated using Pearson's correlation coefficient, with values above 0.7 indicating pairwise correlations that may limit model performance and interpretation (Dormann et al. 2013). When correlation was apparent, the variable with the highest variable importance measures from a single random forest (RF) model, fit with conditional permutation, was retained (Smith et al. 2011). The final list of non-correlated variables was used to fit a full model. Our SDM framework used both Boosted Regression Trees (BRTs) and RFs. BRTs were fitted using the package gbm (Greenwell et al. 2020) with a tree complexity of two, a learning rate of at least 0.01, and at least 1500 trees. RFs were fitted using the package *extendedForest* (Smith et al. 2011) with a maximum of 2000 trees. Each model formulation was fitted 100 times in a bootstrapping approach using randomly selected training data (67%) and withheld evaluation data (33%). Model validation was undertaken by comparing predictions from models tuned using the training dataset with observations from the withheld evaluation data set using the Area Under the Receiver Operating Curve (AUC) and the True Skills Statistic (TSS) metrics (Allouche et al. 2006). AUC scores above 0.7 and TSS scores above 0 indicate adequate model performance (Allouche et al. 2006; Elith et al. 2010). The mean and standard deviation of the AUC and TSS values were calculated across all 100 bootstraps and were used to determine the performance of each model.

Spatial predictions of habitat suitability index (HSI) throughout the study area were generated using gridded environmental data at 250 m x 250 m cell resolution. Three time periods were used to compare changes in bottlenose dolphin habitat suitability over time: June 1994–May 2004, June 2004–May 2014, June 2014 – May 2024, and over a selection of six months: February, April, June, August, October, December to capture any seasonal patterns in occurrence. Thus, we generated 18 predictions of bottlenose dolphin distribution to account for spatiotemporal variability in habitat use. The temporally dynamic variables were averaged across each of the three time periods for each month to generate monthly averaged prediction data frames that were merged with the static environmental data sets. Spatial predictions of HSI were generated for each of the 100 model iterations and averaged to provide mean HSI layers for the BRT and RF models. The standard deviation (SD) of the 100 spatial predictions was calculated as a measure of spatially explicit uncertainty. To minimise the need for inference from a single modelling framework (Robert et al. 2016), ensemble predictions were generated following Anderson et al. (2016), by weighting predictions from the BRT and RF by their model performance statistics (mean AUC). Predictions of habitat suitability were exported as GeoTiff raster layers for the RF, BRT and Ensemble predictions.

The importance of the environmental predictor variables to an SDM can provide insights into the habitat preferences of the species (Brough et al. 2023). Thus, the relative importances of each environmental predictor were calculated from the RF and BRT models using a standardised calculation of the variable importance measure. The importance of each environmental variable predictor p in a RF model, R_p^2, is given by (Ellis et al. 2012):

$$R_p^2 = \frac{R^2 I_p}{\sum_{p'} I_{p'}}$$

where I_p is the accuracy importance of each predictor in a forest, and R_p^2 is the proportion of variance explained by the forest. The goodness of fit, R_p^2, is partitioned among the predictors in proportion to their accuracy importance, I_p. The accuracy importance (I_p) is standardised by the densities across the raw importance from each split in each tree (for each variable *p*) and normalised such that they sum to R_p^2 (Ellis et al. 2012). For BRT models, the predictor variable importance was based on the number of times the variable was selected for splitting, weighted by the split's squared improvement to the model, and averaged over all trees (Friedman & Meulman 2003). The relative importance for both model types were scaled to be a percentage of the influence of all the variables.

Ensemble partial dependence plots of the relationship between the environmental variables and the relative habitat suitability for bottlenose dolphin were produced by generating weighted predictions across the full gradient of each environmental variable retained in the final model, using the respective AUC evaluation scores for each model type as weights. Uncertainty around the weighted response was summarised by the weighted mean uncertainty (SD) across each model type. Partial dependence plots provide an understanding of the nature of the relationship between environmental variables and dolphin occurrence, and thus help to determine the features of key habitats.

Present 'snapshot' of distribution

A 'snapshot' of current distribution patterns for bottlenose dolphins in the Bay of Islands, using data gathered during our surveys, was generated using Kernel Density Estimation (KDe; Worton 1989). Following Brough et al. (2019) and Bennington et al. (2023), KDe were fit using effort weighted point estimates of relative abundance from group size estimates of encounters with dolphin groups. All seventeen encounters were used as input data, with the starting X and Y coordinates being used as the group's location and the mean group size estimate provided a value for relative abundance. Each group size estimate was standardised by the total kilometres of survey effort contained within the 5 km x 5 km cells of the study area (Figure 3-2) to represent the number of dolphins per km survey effort. A fixed KDe was calculated in ArcGIS Pro (version 3.1.3, ESRI), using the default method to calculate smoothing bandwidth (Seaman & Roger 1996) (i.e., using the standard deviation of X and Y coordinates). Spatial interpolation included a barrier for the coastlines and islands of the study are to prevent interpolation across land (Brough et al. 2019). The spatial density surface was created using a 100 x 100 m resolution spatial raster layer, and reveals the current core areas for bottlenose dolphins in the Bay of Islands for the most recent year. This analysis provides an opportunity to contrast distribution patterns with the more long-term averages calculated via the SDMs.

2.4.3 Acoustic analysis

CPOD acoustic loggers record the incidence and characteristics of echolocation-like pulses without recording raw acoustic data. Customised algorithms are then fit to the characteristics of acoustic pulses to classify odontocete echolocation click trains (e.g., regularly spaced signals with similar characteristics) (Nuuttila et al. 2013; Roberts & Read 2015). The KERNO classifier was used in CPOD.exe (version 2.048) to detect click trains that originate from cetaceans and vessel echo-sounders. KERNO then attributes a quality score to each train to specify the likelihood of a train originating from the acoustic source of interest ('high', 'medium', 'low', 'doubtful'), based on the signal's amplitude, duration, frequency and inter-click interval (Nuuttila et al. 2013). The retained click trains are then further classed into four groups based on the train's acoustic characteristics: NBHF (narrow band high frequency clicks), Other cet (non-NBHF dolphins), Sonars (vessel echosounders) and Unclassed.

Bottlenose dolphins generate broadband, short duration echolocation clicks with a peak-to-peak source level up to 228 dB re 1 µPa at 1 m, with peak frequencies between 30 and 150 kHz and duration between 8 and 72 µs (Au 2012; Au et al. 2012). After manual screening of 10% of detected trains to discern alignment of the click train characteristics with those reported for bottlenose dolphins, we retained all 'high' and 'medium' click trains. Using the export functions within CPOD.exe, we calculated two metrics to summarise the relative use of each deployment station by bottlenose dolphins and vessels: 1) Detection positive minutes (DPM) – a routinely used index of relative presence of echolocating species (Nuuttila et al. 2013; Brough et al. 2020); and 2) the proportion of days with positive detections. Each metric was summarised for click train classes 'Other cet' which retained signals that match bottlenose dolphins and 'Sonar' as an indication on the relative presence of vessels.

2.4.4 Vessel compliance and dolphin interactions

Vessel compliance was assessed according to the rules of the BOIMMS. For the marine mammal safe zones, vessels were termed non-compliant if their estimated speed exceeded 5 knots. Anywhere within the BOIMMS (i.e., within and outside of the marine mammal safe zones), a vessel was termed non-compliant if it failed to stop its engine or drop its sails within 300 m from dolphins until the dolphins were more than 300 m away. Further, swimming with dolphins was termed non-compliant anywhere within the sanctuary.

Vessel compliance and dolphin interactions were summarised and presented through descriptive plots, showing number of interactions and the percentage of vessel compliance.

3 Results

3.1 Summary of fieldwork

The study area was surveyed on 20 discrete days between 29 March and 25 May 2024. Survey duration ranged from 7 to 8.38 hrs (median = 8.15 hrs). During these discrete surveys, compliance with vessel speed restrictions within the two safe zones was assessed on 24 occasions, with each safe zone being assessed for 12, 20-minute intervals. There were 17 encounters with bottlenose dolphins on 13 discrete survey days. Dolphins were encountered during 65% (n = 13) of the 20 survey days and 0.85 groups were sighted per survey day.

Systematic survey effort for bottlenose dolphins covered the BOIMMS, with higher effort in areas that were in-transit from daily start location (Waitangi/Opua) and around the marine mammal safe zones (Figure 3-2). The few areas of lower survey effort were in the upper reaches of the shallow arms of the major inlets (e.g., Kerikeri, Waikare), that are not regularly used by dolphins (Hartel et al. 2014, Peters & Stockin 2016, Guerin 2022). A total of 1,994 km of 'on-effort' distance was carried out across the study area between 29 March and 23 May 2024.

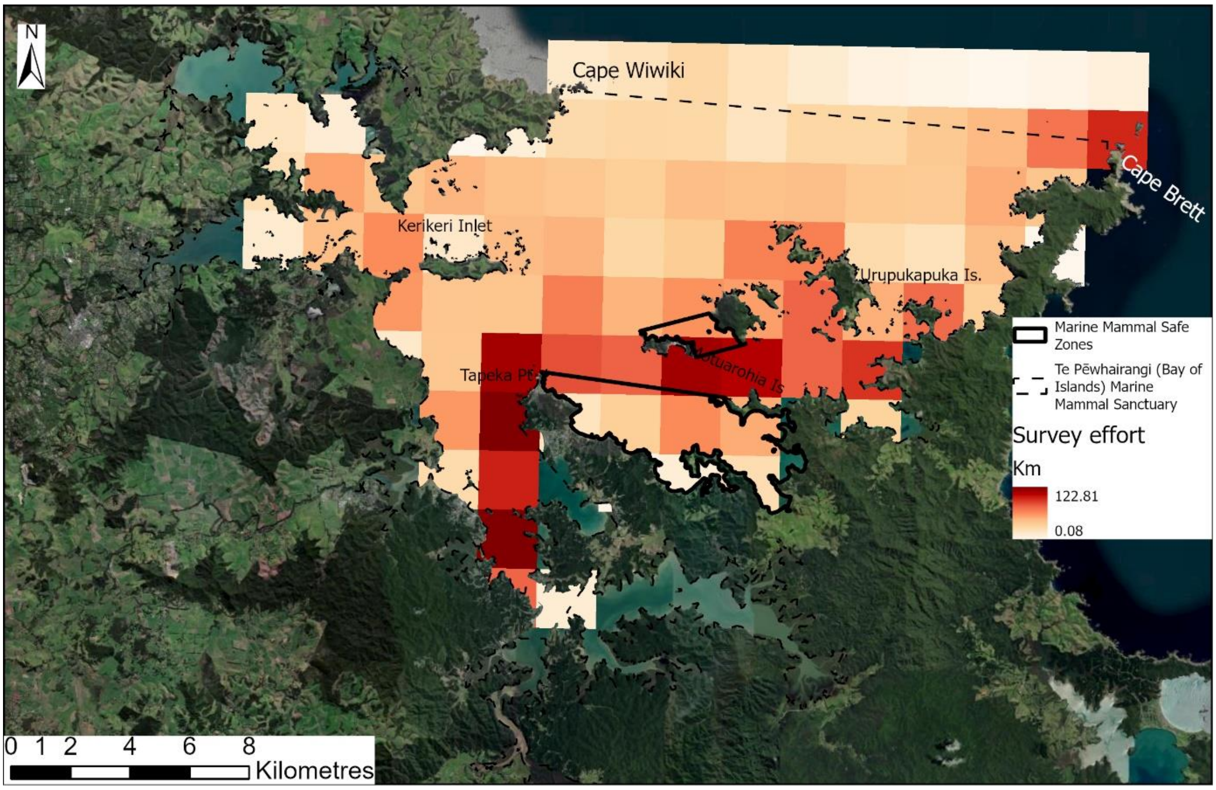

Figure 3-2: Survey effort. The distribution of survey effort recorded during systematic surveys for bottlenose dolphins in the Bay of Islands.

3.2 Photo-identification results

A total of 17 dolphin groups were encountered on 13 discrete (65%) survey days. Single groups were the most common occurrence in the study area on any given day accounting for 45% of survey days (n = 9). On 4 survey days, the same groups were encountered twice (but in different locations) and were omitted from the photo-identification analysis. Thus, only a single discrete group of dolphins was approached for photo-ID on each of the 13 days that yielded encounters. Group size ranged from 2 to 30 individuals (median = 4).

All identifiable individuals were associated within a single network cluster (Figure 3-4), however there were two sub-groups present during our surveys with particularly strong associations between eleven individuals that were most frequently sighted (Figure 3-4). These individuals were the most frequent users of the BOIMMS during our surveys but were also seen regularly with individuals that were seen sporadically (Figure 3-4).

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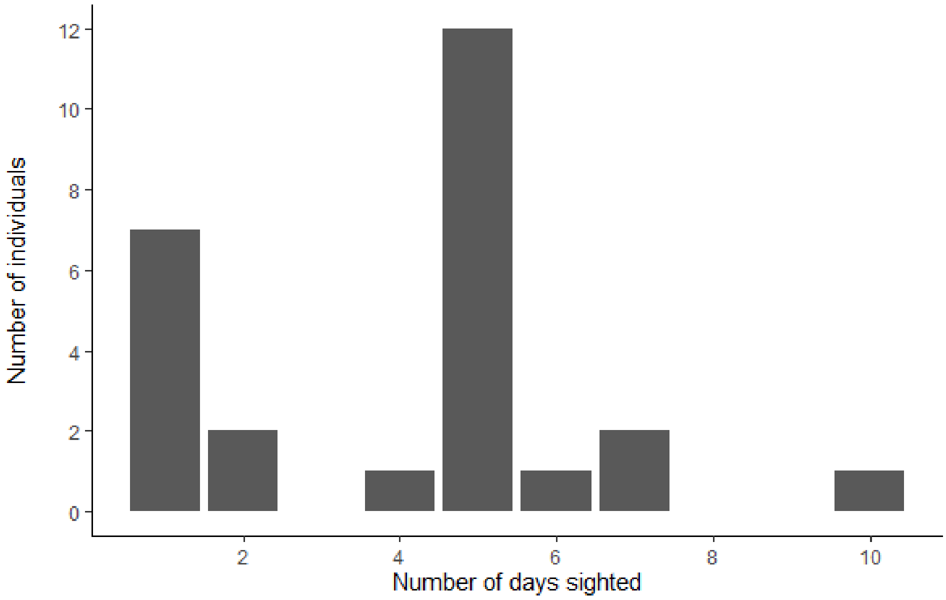

Figure 3-3: Resighting frequency. Number of days on which each individually identifiable bottlenose dolphin was sighted within the Te Pēwhairangi (Bay of Islands) Marine Mammal Sanctuary between March 29 and May 23, 2024.

There were 105 encounters with 24 identifiable individuals (median = 5). Of these individuals, 73.1% (n = 19) were sighted on more than one day (range 1-10, Figure 3-3). Calves were observed during 38.5% of encounters (n = 5). A total of four different calves were documented through repeated association with distinct adult dolphins.

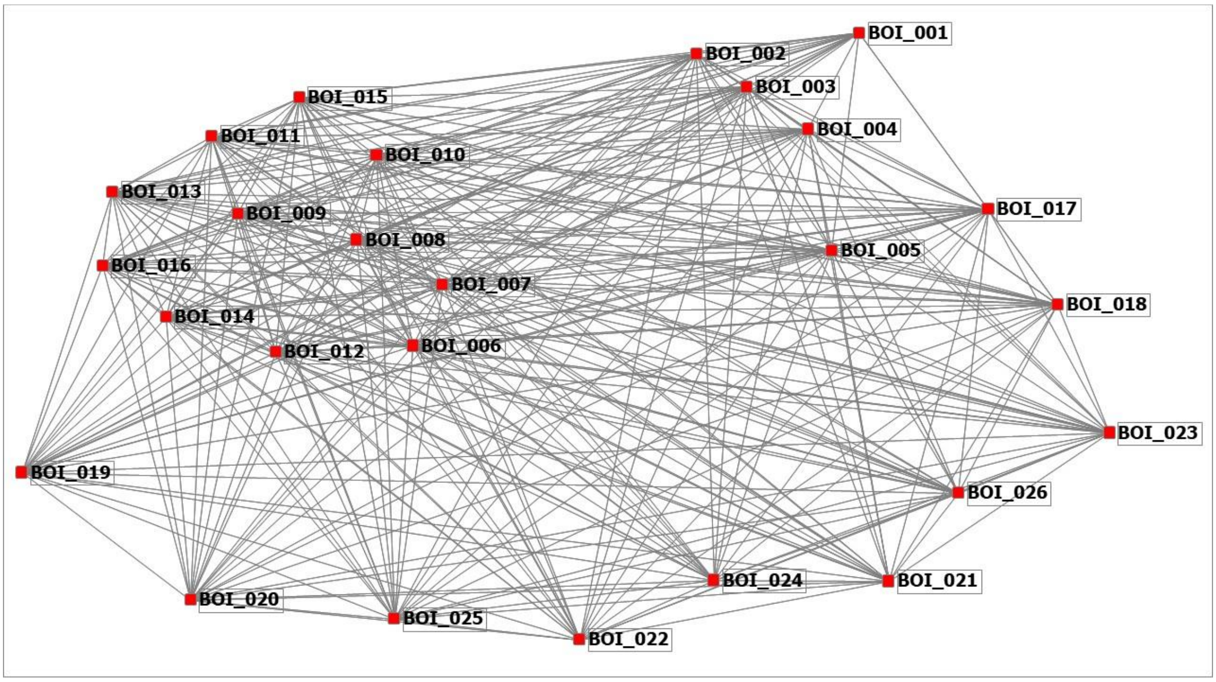

Figure 3-4: Social network diagram. Social network diagram of 26 bottlenose dolphins photo-identified between 29 March and 23 May, 2024, using a spring-embedded layout. Individual dolphins are represented by red nodes with their corresponding catalogue number.

3.3 Data integration

A key objective of this project was to integrate data collected under the fieldwork described above with existing datasets held by several researchers. The following databases were integrated into a master photo identification database (after filtering according to sections 2.1.7 and 2.3 of this report):

- University of Auckland, 1993-2012, 465 individuals, (353 after filtering for quality/distinctiveness).
- Massey University, 2012-2015 and TriOceans, 2017-2022, 239 individuals, (150 after filtering for quality/distinctiveness and removal of oceanic individuals). The two datasets were combined as they use the same catalogue numbers.
- Far Out Ocean Research Collective/NIWA, 2024, 26 individuals, (24 after filtering for quality/distinctiveness).

In total, 128,031 photos were matched, resulting in 631 distinct individuals. These 631 individuals were then matched to the Far Out Ocean Research Collective Oceanic Bottlenose Catalogue (708 individuals), with 446,778 images matched, resulting in 48 matches (all from the Massey University/TriOceans data sets) which were excluded from the dataset along with any other distinct individuals that occurred in the same encounters. The final master photo identification dataset includes 540 *distinctive* or *very distinctive* coastal ecotype individuals with images of *good* or *excellent* quality.

Surveys were carried out on 1,605 discrete days, with photo-identification of the 540 individuals mentioned above occurring on 1345 days between 1993 and 2024. Overall, there were 1,317 recorded encounters with dolphin groups. The discrepancy between encounter, photo-ID and effort data are largely due to missing encounter data that was not available in the long-term datasets reviewed by this study.

3.4 Updated population demographics

3.4.1 Local population abundance - snapshot

A total of 24 unique individuals were included within the ‘mark’ period and 16 unique individuals were sighted in the recapture period. The closed MR model provided estimates for probability of capture of 0.47 (CI = 0.14–0.83), probability of recapture of 1 (CI = 1–1), and the number of individuals not seen of 9.75 (CI=1.4–65.29) (Table 3-4).

The derived parameter estimate for the abundance of marked individuals in the local population between March and May 2024 is 36 (CI = 27–91). A mark ratio of 0.923 was calculated using random photographic techniques during dolphin encounters. Thus, the final snapshot local population estimate for bottlenose dolphins in the Bay of Islands as of May 2024 is 39 marked and unmarked (adult) individuals (CI = 29–99).

A probability of recapture of 1 is possible as all individuals from the ‘mark’ period were resighted in the single recapture period. High upper confidence limits for the parameters f0, and subsequently N̂, and N are due to the high uncertainty in the estimate for *p.* In turn, this uncertainty is due to a relatively high proportion of individuals (38%) being first encountered in the recapture period, and an inability for the closed model formulation to specify a time-varying *p* (Lukasc 2009).

Table 3-4: Mark-recapture parameter estimates. Parameter estimates for closed mark-recapture analysis of population abundance. Model parameters are probability of capture (p), probability of recapture (c), number of individuals not seen (f0). Derived parameters are population abundance for marked individuals (N̂) and population abundance for marked and unmarked individuals (N).

Parameter	Estimate	se	lcl	ucl
p	0.47	0.22	0.14	0.83
c	1.00	0.00	1.00	1.00
f0	9.57	12.15	1.40	65.29
N̂	36	12	27	91
N	39	13	29	99

3.4.2 Long term trends in population demographics

The data used for estimating long term population demographics using the robust design framework was characterised by overdispersion (median Ĉ = 3.2, SE = 0.384), suggesting extra-binomial variation in the data. Thus, models were adjusted by inflating the variance-covariance matrix of the estimated parameters and QAIC was used for model selection by specifying the ‘chat’ argument in the model specification.

The most parsimonious robust design model for estimating the long-term population demographic parameters had:

- Constant, transience-informed apparent survival ϕTr(.),

- Probability of capture/recapture varying by primary period (*p*Mix(t)*),*
- Constant mixture probability of latent classes π(.),
- The number of individuals not sighted as varying by primary period (*f0*(t))
- Markovian temporary emigration with γ′ varying by decadal time period and γ″ being constant (γ′(tp) ≠ γ″(.))

This model was supported by a high weight of evidence (model weight= 1, the next best model: ΔQAICc of 59). For this reason, there was no need to consider model averaging approaches for parameter estimation (Burnham & Anderson 2000). All parameter estimates are for marked individuals only with the exception of the derived estimates of abundance that have been scaled by mark ratio.

Table 3-5: Robust design model selection table. List of the top 9 (of 48) candidate model formulations for the estimation of population demographic parameters using the long-term photo identification dataset. Parameters include ϕTr (apparent annual survival considering transience), γ′ ≠ γ″(Markovian emigration), γ′ = γ″(random emigration), π (latent class mixture probability), *p*Mix (capture probability considering heterogeneity), f0 (n individuals not captured). (t) indicates a parameter varying by time (primary period), (tp) indicates variation by decadal time period, (.) indicates a constant parameter. Models are ranked by Quasi-likelihood Akaike Information Criterion (QAIC), delta QAICc (the difference between QAIC and the lowest QAIC) and model weight.

Model	df	QAICc	ΔQAICc	weight
φTr(.) + γ′(tp) ≠ γ″(.) + π(.) + pMix(t) + f0(t)	148	1593	0	1
φTr(.) + γ′(tp) ≠ γ″(tp) + π(t) + pMix(t)+ f0(t)	196	1652	59	0
φTr(.) + γ′(tp) ≠ γ″(.)+ π(t) + pMix(t) + f0(t)	194	1653	60	0
φTr(.) + γ′(.) ≠ γ″(tp) + π(t) + pMix(t) + f0(t)	192	1657	64	0
φTr(.) + γ′(.) ≠ γ″(tp) + π(t) + pMix(.) + f0(t)	100	1697	104	0
φTr(.) + γ′(tp) ≠ γ″(tp) + π(t) + pMix(.) + f0(t)	104	1697	104	0
φTr(t) + γ′(tp) ≠ γ″(.) + π(.) + pMix(t) + f0(t)	238	1706	114	0
φTr(t) + γ′(.) ≠ γ″(tp) + π(.) + pMix(t) + f0(t)	236	1707	114	0
φTr(.) + γ′(tp) ≠ γ″(.) + π(t) + pMix(.) + f0(t)	102	1707	114	0
φTr(.) + γ′(.) ≠ γ″(tp) + π(t) + pMix(.) + f0(t)	102	1724	131	0
φTr(t) + γ′(.) ≠ γ″(tp) + π(.) + pMix(t) + f0(t)	238	1728	135	0
φTr(.) + γ″(tp) = γ′(tp) + π(.) + pMix(t) + f0(t)	147	1753	160	0

Apparent annual survival for resident bottlenose dolphins (i.e., those sighted over more than a single primary period) was 0.90 (95%CI = 0.88-0.91), while apparent survival for the first primary period of sightings (i.e., including transient individuals) was 0.67 (95%CI = 0.57-0.75). Capture and recapture probabilities were different between the two latent classes, with class 1 having a range of *p* from near 0 to 0.988 (median = 0.324) and class 2 ranging from near 0 to 0.998 (median = 0.358). Capture probabilities were lower in the early period of the study and became increasingly higher post 2010.

Temporary emigration was stable across the first two decades of the study period, with the probability of remaining outside of the Bay of Islands between primary periods (γ′) being estimated at 0.81 (95%CI = 0.75–0.86) and 0.80 (95%CI = 0.74–0.85) for the time periods between 1997 and 2003 and 2004 and 2013 respectively (Figure 3-5). Υ′ increased significantly in the late time period

(2014–2024) to 0.93 (95%CI = 0.90–0.95). The probability of moving outside of the Bay of Islands from one primary period to the next was stable over the study period at 0.143 (95%CI = 0.123–0.165) (Figure 3-5).

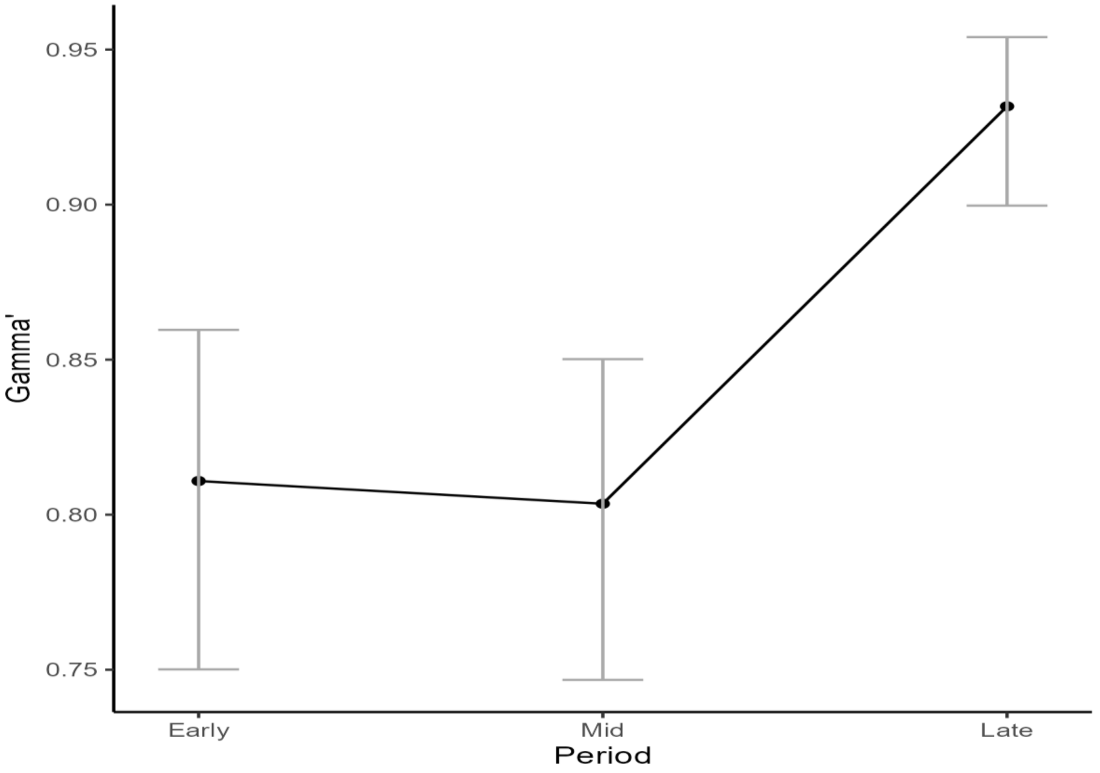

Figure 3-5: Temporary emigration. Rates of Markovian temporal emigration for three decadal time periods over the full study period between 1997 and 2024; early (1997–2003), mid (2004–2013), and late (2014–2024). Gamma' (denoted as γ' in the text) is the probability of remaining outside of the study area between primary periods. Error bars are 95% confidence intervals.

Estimates of local population abundance of marked and unmarked individuals suggest a significant decline in abundance over the study period from 1997 to 2024 (Figure 3-6). Local abundance was highest in summer of 1997 with an estimate of 244 individuals (95%CI = 116–515) and was lowest in summer 2019 with a local population abundance estimate of 16 individuals (95%CI = 16–16). There were periods of relative stability in local population abundance between 2005 and 2015, but steep declines in abundance between 1999 and 2003, and between 2014 and 2015 (Figure 3-6). Other than a single increase in abundance in the summer of 2021, there have been no obvious signs of local population recovery since the 2015 decline. The current local abundance of the population (as of 2024) is 40 (95%CI = 27–58) using the robust design outputs.

Pooling the primary period abundance estimates by season reveals a decrease in local abundance has occurred across all seasons (Figure 3-7). In general, historical abundance has been higher in summer and autumn seasons and lower in winter. Thus, the decline in summer/autumn abundance has been most pronounced. It should also be noted that there has been insufficient photo-identification effort for analysis using the robust design during summer periods since 2021. Thus, the current state of local abundance during this important time of the year is unknown.

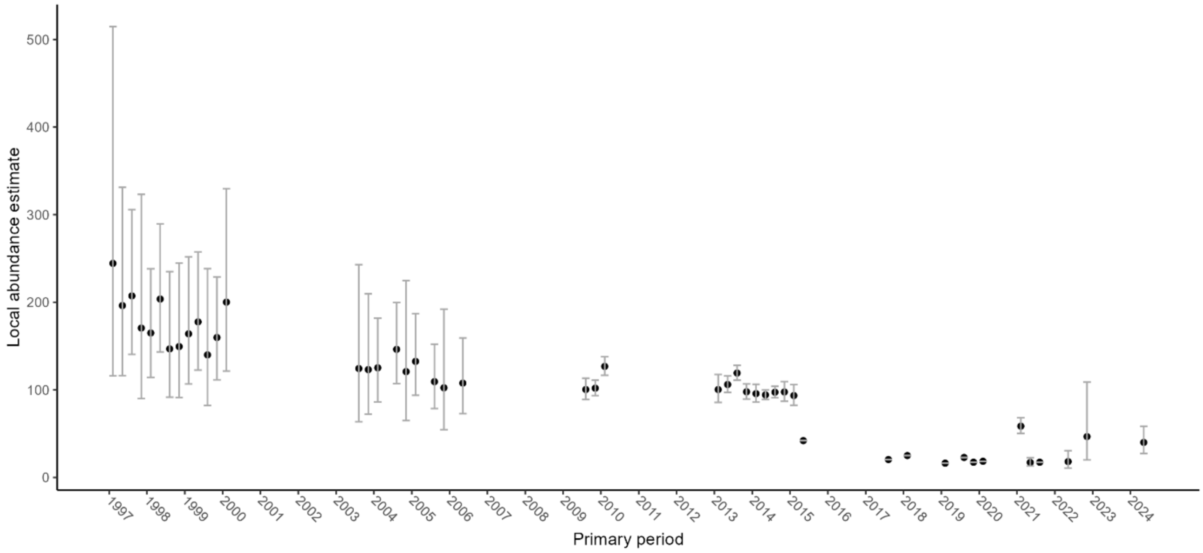

Figure 3-6: Local population abundance by primary period. Estimates of local population abundance (of both marked and unmarked individuals) for 47 primary periods derived from robust design mark-recapture models. Error bars a 95% confidence interval. Note the gaps where there were not sufficient photo-identification data for modelling.

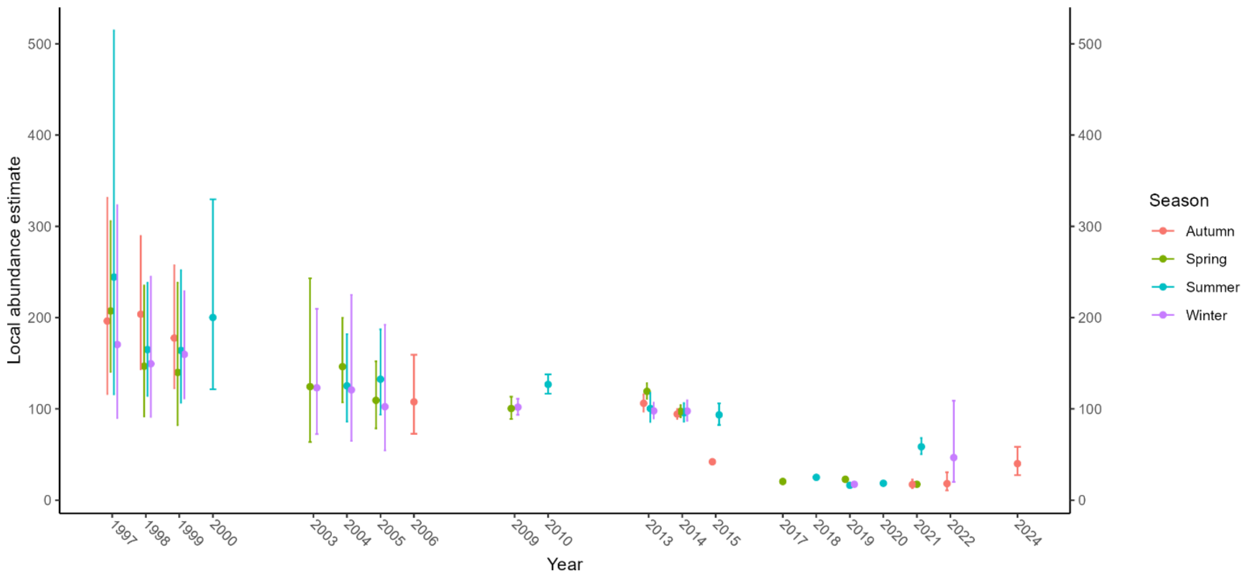

Figure 3-7: **Seasonal abundance estimates.** Local abundance estimates (for marked and unmarked individuals) separated by the season represented by each primary period. Error bars are 95% confidence intervals.

Apparent calf mortality

Calf mortality rates have been highly variable over the study period and were characterised by high variance using the Wilson's score to approximate a 95% confidence interval. First year apparent calf mortality fluctuated from 1 (i.e., all calves died) to 0 (all calves survived) (Figure 3-8). There was considerable variability in first year apparent mortality rates between 1994 and 1999, with a period of low mortality (0) in 2001 and 2002, though it should be noted that research effort was particularly low over these years. There was a gap in the available data on the fate of known calves between 2005 and 2012, thus first year survival could not be estimated for this period. Apparent annual first year calf mortality rates appear to have improved for the most recent years (2017-2022) with all calves having survived their first year with the exception of 2 calves born in 2018 (Figure 3-8). However, only a single calf of known fate was observed for each year over this period (except 2018), and research effort was also low.

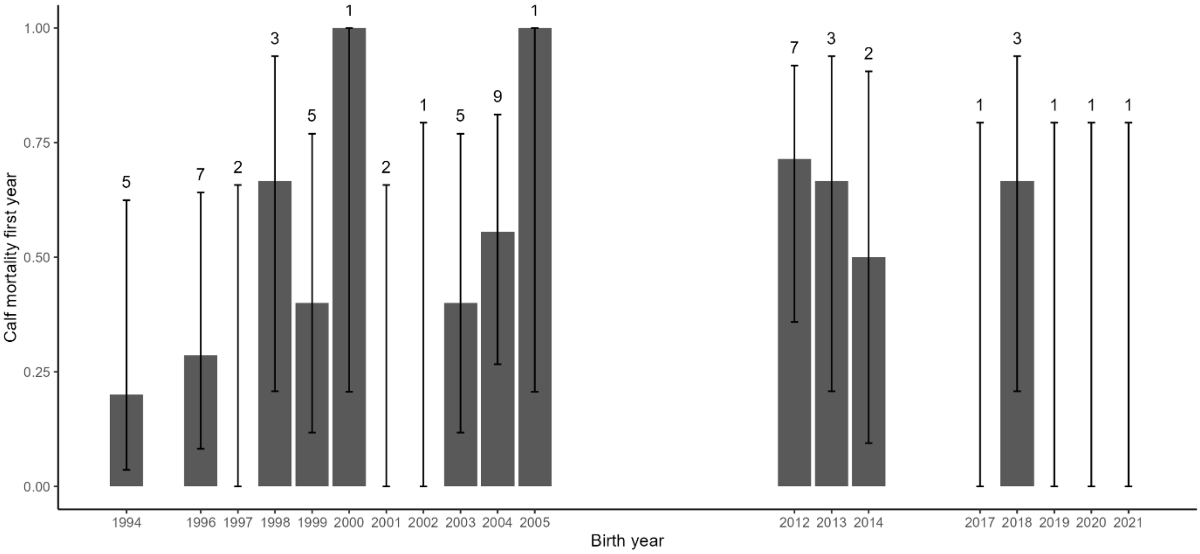

Figure 3-8: Calf mortality in the first year of life. Bar height indicates the proportion of calves known to have died within the first year of life divided by the number of calves born that year with known fate. The number above the error bar indicates the number of calves born that year with known fate. The error bars are the Wilson Score intervals of calf mortality based on the binomial variance estimate. Years represent reproductive seasons, i.e., 1994 spans July 1994 through June 1995.

Apparent second year calf mortality rates were generally lower than those estimated for the first year of life, with all 1-year-old calves dying in three years (1997, 2009 and 2013). Years with all calves surviving the second year included 1996, 1998–2001, 2012, 2014 and for a recent period between 2017 and 2020.

For 1994-2006, Tezanos-Pinto et al. (2015) found that, of the 41 calves for which fate could be determined, 24 calves survived into their first year, and 17 calves did not, giving an overall calf mortality of 0.41 (95% CI = 0.26 – 0.57) for the first year of a calf's life. In this study, of the 19 calves for which fate could be determined after 2006, it was found that nine calves survived into their first year, and 10 did not, giving an overall calf mortality of 0.53 (95% CI = 0.3 - 0.75) for the first year of life, an increase in mortality rate compared to Tezanos-Pinto et al. (2015).

Tezanos-Pinto et al. (2015) found that of 19 calves whose fate could be determined into their second year of life, five died before two years of age, giving an overall calf mortality of 0.26 (95% CI = 0.07 – 0.46) for the second year of a calf's life. In this study, post 2006 the fate of 10 calves could be determined into their second year of life, of which three died before two years of age, giving an overall calf mortality of 0.3 (95% CI = 0.02 – 0.58) for the second year of life, which is similar to that generated by Tezanos-Pinto et al. (2015).

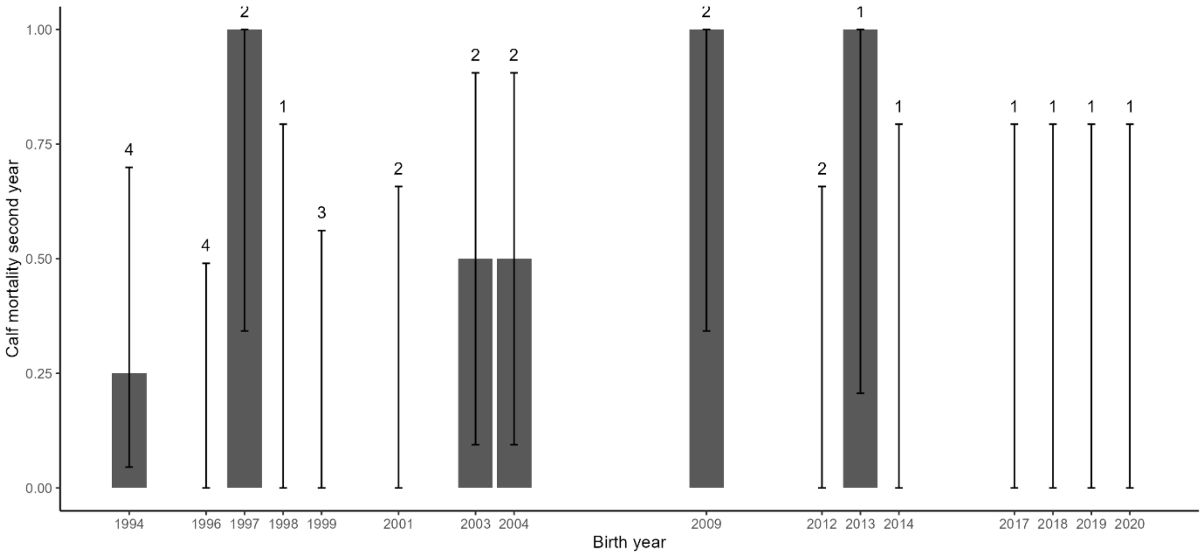

Figure 3-9: Calf mortality in the second year of life. Bar height indicates number of calves known to have died within the second year of life divided by the number of calves that survived into their first year, with known fate. The number above the error bar indicates the number of calves born that year with known fate. The error bars are the Wilson Score intervals of calf mortality based on the binomial variance estimate. Years represent reproductive seasons, i.e., 1994 spans July 1994 through June 1995.

Reproductive rate

A total of 72 individuals were identified as reproductively active females throughout the study period. Reproductive rates in the Bay of Islands were variable over the study period and range from a minimum of 0.04 in 1998 and 2001 and a maximum of 0.4 in 2018 (Figure 3-10). Reproductive rate was lower during the early period of research in the Bay of Islands (1994–2003) compared to later (2012–2022). Peaks in reproductive rate occurred in 2004, 2012 and 2018. It should also be noted that the number of reproductive females in the local population has steadily declined over time, coincident with the overall decline in the population.

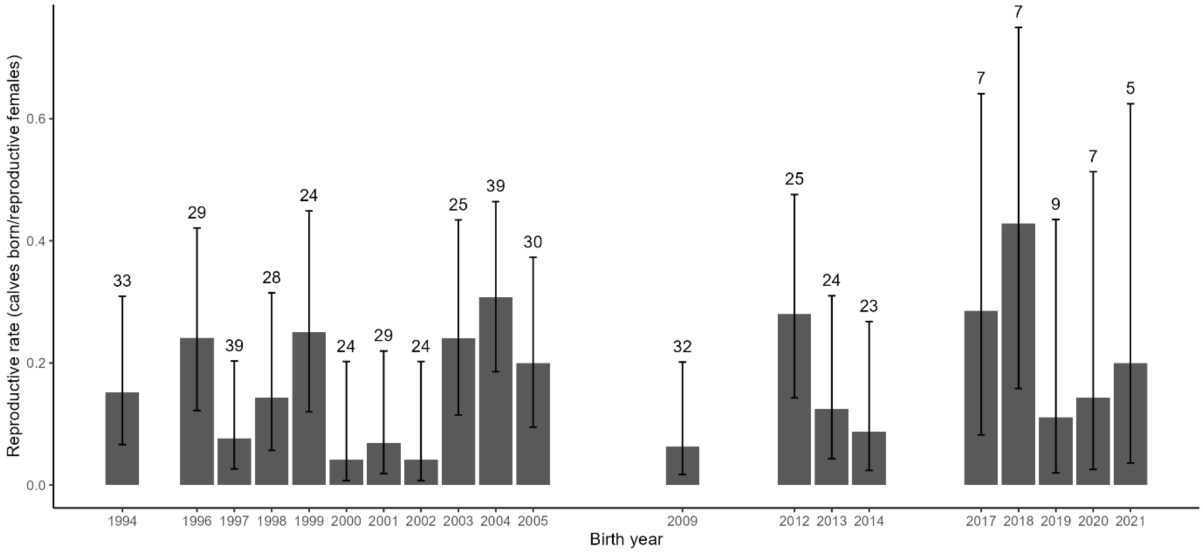

Figure 3-10: Reproductive rate. Bar height indicates number of calves born that year divided by the number of reproductive females for the corresponding year. The number above the error bar indicates the number of reproductive females for that year. The error bars are the Wilson Score intervals of reproductive rate Years represent reproductive seasons, i.e., 1994 spans July 1994 through June 1995.

3.5 Spatiotemporal distribution

3.5.1 Dynamic species distribution modelling

The full range of environmental variables were able to be included in the SDMs with the exception of mixed layer depth (MLD) which was highly correlated with bathymetry (Pearson's correlation = 0.9). MLD had a lower importance score in the RF model fit with conditional permutation and was thus excluded as an environmental predictor variable. The dynamic SDM framework generated robust predictions of bottlenose dolphin distribution throughout the BOIMMS, with mean AUC evaluation scores of 0.78 (SD = 0.02) and 0.74 (SD = 0.02) for the RF and BRT, respectively and TSS scores above 0 (Allouche et al. 2006).

Table 3-6: Species distribution model evaluation. SDMs evaluated by area under the received operating curve (AUC) and true skill statistic (TSS), for both the training (train) and evaluation (eval) datasets and models (RF, Random Forest; BRT, Boosted Regression Trees). Values represent mean scores (and associated standard deviation) across 100 model runs evaluated with the training (.train) and withheld (.eval) data.

	RF mean	RF SD	BRT mean	BRT SD
AUC.train	0.88	0.01	0.75	0.02
TSS.train	0.61	0.03	0.45	0.03
AUC.eval	0.78	0.02	0.74	0.02
TSS.eval	0.43	0.03	0.36	0.03

There were notable seasonal differences in predicted distribution (measured by habitat suitability index, HSI) for bottlenose dolphins in the Bay of Islands. HSI was highest in the months of December, February and April and lowest in the month of August across all periods (Figure 3-11), revealing habitat is less suitable during the cooler months of the year. Further, the areas of highest HSI during the cooler months (June–October) tended to be more bound to sheltered waters around the inner Islands on the eastern side of the Bay. In contrast, areas of high relative importance in the warmer months were spread more widely and in particular, a hotspot in the triangle between Tapeka Point, the Black Rocks, and Motuarohia Islands was evident. These trends were consistent across each of the three time periods (Figure 3-11).

In general, habitat suitability in the Bay of Islands has increased over time and there was limited variability in the locations of importance for bottlenose dolphins over the 30-year study period (Figure 3-12). Compared to the early period, areas within the inner Islands have become more suitable in the two latter periods during June, the area around Tapeka Point is more suitable during October and December in both latter periods and an area seaward of Motuarohia and Moturoa Islands has become more suitable habitat over time. Few locations experienced a decline in habitat suitability over time, the exceptions being the Kerikeri Inlet in August and the ‘middle ground’ west of Okahu Island in April (Figure 3-12).

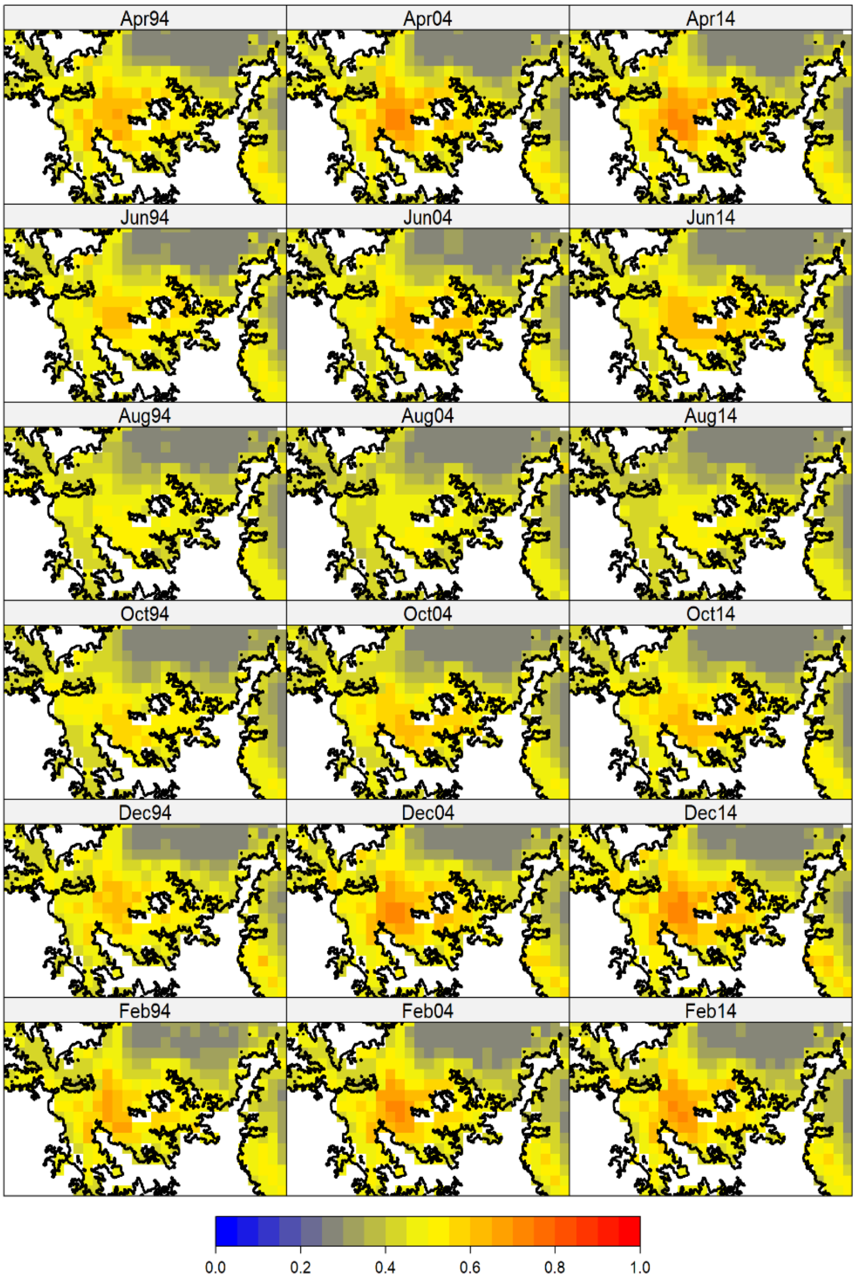

Figure 3-11: Distribution of coastal bottlenose dolphin. Predicted mean habitat suitability index for bottlenose dolphins (0-1 indicated by colour key) from species distribution modelling across the study area for three time periods (columns left to right: 1994–2003, 2004–2013, 2014–2024), for six months (rows top to bottom: April, June, August, October, December, February). See Appendix B for associated uncertainty (SD).

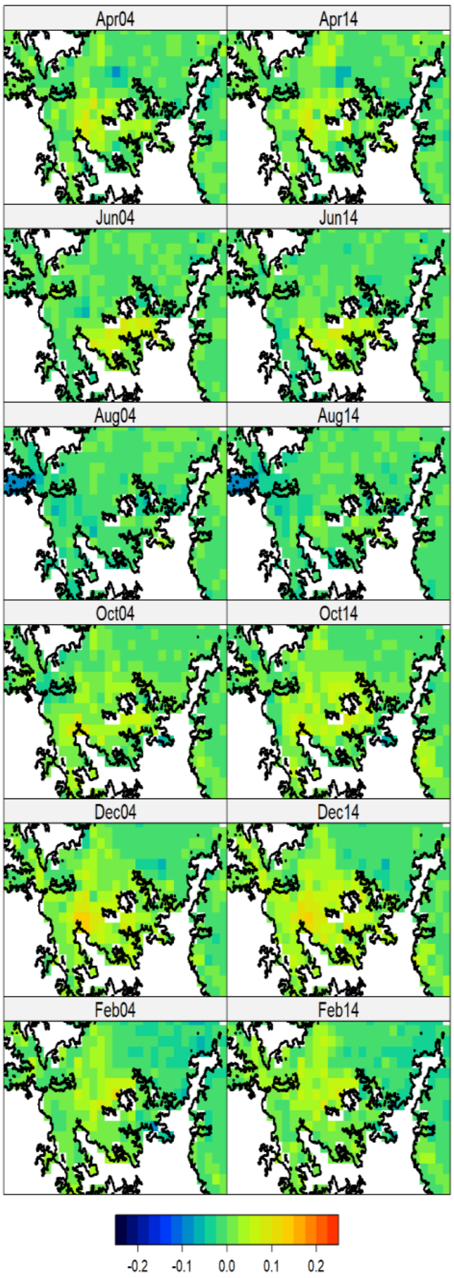

Figure 3-12: Differences in habitat suitability over time. Monthly difference layers created by subtracting the early period HSI from the mid (left column) or late (right column) HSI for each month. Positive values (red side of colour key) indicate an increase in habitat suitability over time.

Bathymetry was the most important variable for predicting the distribution of bottlenose dolphins, contributing 31.14% to the predictive performance of the ensemble model. Habitat suitability declined with increasing depth and was highest at depths around 10 m (Table 3-7 and Figure 3-13). Benthic sediment disturbance (a proxy for swell exposure) also had a high contribution (25.76%), with partial dependence plots suggesting preferences for more sheltered waters (Figure 3-13). The relationship between bathymetric position index (BPI) and habitat suitability indicates a preference for areas with depressions in the seafloor (i.e., basins). Turbidity (i.e., particulate backscatter) made a moderate contribution to the models (6.55%) and showed a negative relationship with HSI – with the highest HSI values in less turbid water conditions. The slope of the seafloor exhibited a negative relationship with HSI, suggesting dolphins prefer areas with minimal seafloor relief. Sea surface temperature (4.4% contribution) had a positive relationship with HSI, with the highest HSI values being predicted between 20 and 24 degrees Celsius. (Table 3-7 and Figure 3-13). Gradient in SST and tidal current velocity made the lowest contribution to the models, with both exhibiting limited variation in HSI across their respective gradients (Figure 3-13).

Table 3-7: Relative importance of environmental variables. The standardised relative importance (% contribution) of environmental variables to the species distribution model, with means and SDs for each model type (RF and BRT) and for the ensemble (ENS).

Variable	RF mean	RF SD	BRT mean	BRT SD	ENS mean	ENS SD
Bathymetry	13.59	1.39	48.70	22.26	31.14	11.83
Seafloor wave disturbance (BedDist)	14.71	1.73	36.81	23.16	25.76	12.45
Bathymetric position index (BPI fine)	9.31	1.02	5.17	5.46	7.24	3.24
Turbidity (BBP)	10.24	0.75	2.85	4.39	6.55	2.57
Month	10.54	0.66	1.95	2.85	6.24	1.76
Chlorophyl *a* concentration (CHL)	9.08	0.79	1.42	2.92	5.25	1.85
Slope of the seafloor (Slope)	8.20	0.66	1.81	3.17	5.00	1.91
Sea surface temperature (SST)	8.09	0.67	0.71	1.28	4.40	0.97
Gradient in SST (SSTGrad)	8.31	0.74	0.31	0.95	4.31	0.84
Tidal current velocity (TC)	7.94	0.56	0.26	0.65	4.10	0.60

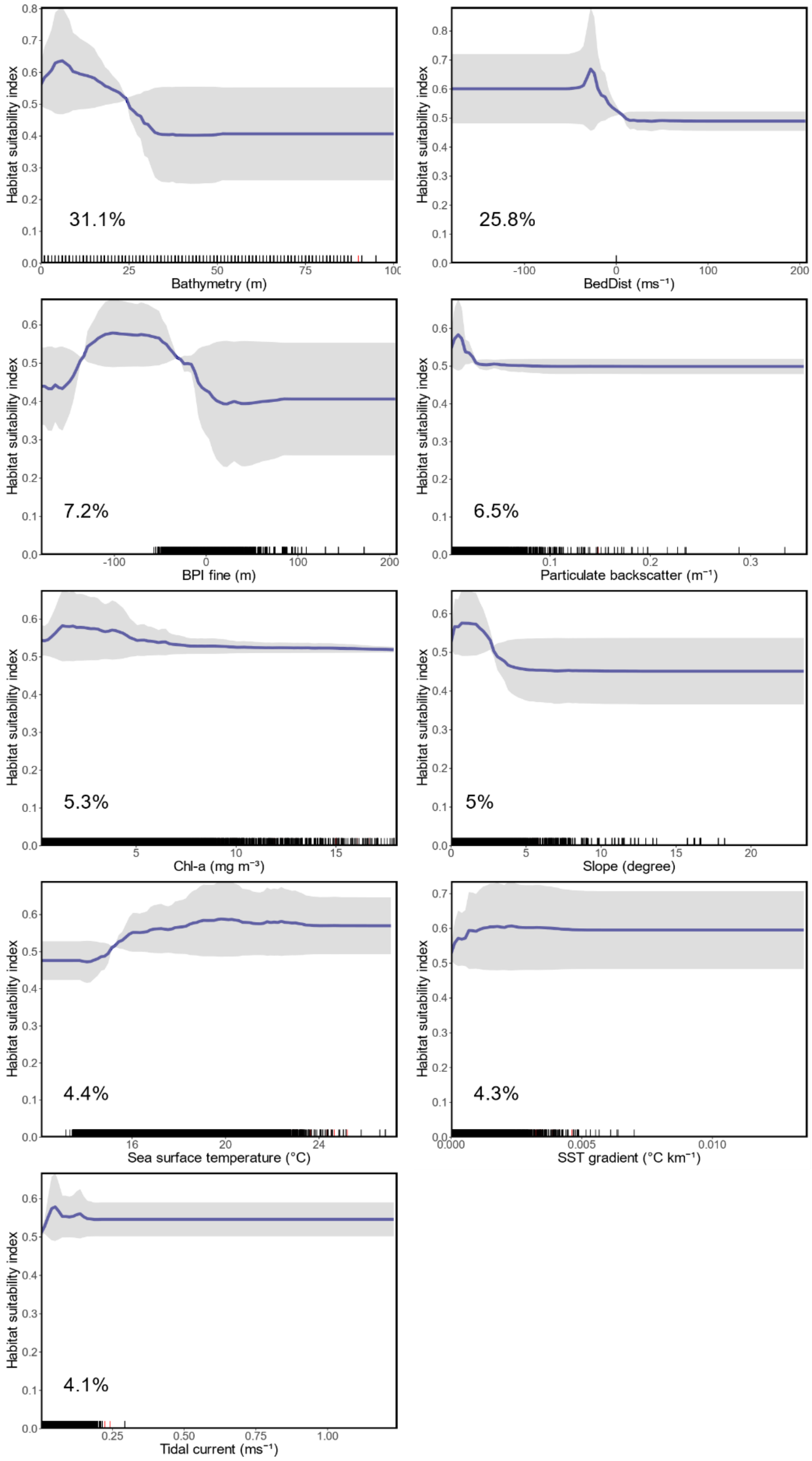

Figure 3-13: Partial dependence plots. The relationship between the habitat suitability for bottlenose dolphins and environmental variables in the Bay of Islands. Plots are generated from the final ensemble model and therefore represent the weighted average relationship across the two model types (BRT and RF). The

rugplot above the x-axis represents the distribution of occurrence records across the environmental gradient.

Current distribution patterns

Encounters with bottlenose dolphins collected during surveys conducted during March to June 2024 almost exclusively occurred within or nearby the ‘middle ground’, especially between Motuarohia Island and Moturoa Island at the mouth of the Kerikeri Inlet (Figure 3-14). Several encounters occurred around the Urupukapuka/Waewaetoria/Okahu Island complex.

The effort-weighted kernel density estimate highlighted a core area of high relative density for bottlenose dolphins between Motuarohia and Moturoa Islands (Figure 3-14). An area of very high density occurred in the middle ground off Okahu Island and is attributable to a single sighting with a high group size estimate (30) compared to the median (4), which happened to overlay an area with comparatively lower effort (Figure 3-2, Figure 3-14). There were no encounters in either of the marine mammal safe zones, which correspondingly had moderate/low relative density in the kernel density estimate. This ‘snapshot’ of current distribution patterns for the Autumn of 2024 agrees closely with the SDM based predictions for the same season/time period (Figure 3-11).

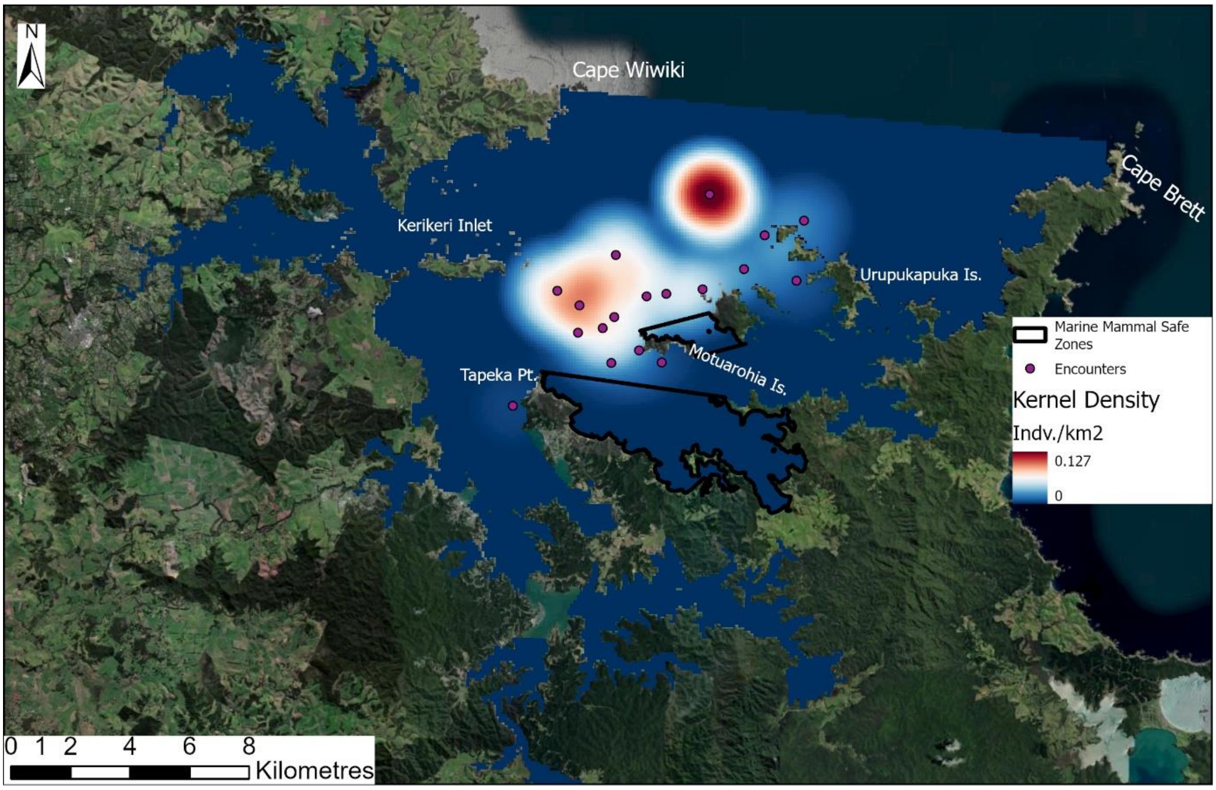

Figure 3-14: Spatial distribution of bottlenose dolphins. Encounters with bottlenose dolphins made during systematic survey of the Bay of Islands overlaid upon weighted kernel density estimation to determine core areas.

3.6 Acoustic habitat use

Four of the five CPODs were successfully retrieved and had data downloaded in May 2024. The fifth, deployed near Waewaetorea passage, was noted as missing in early May. Subsequent searches and contact with regular users and the harbourmaster have failed to reveal its whereabouts. Given the sheltered location (i.e., low likelihood of mooring failure), it has most likely been appropriated illegally. We will continue to search the area in the hope that it surfaces.

The remaining four CPODs logged a total of 5,232 hours of acoustic data across the four locations. Between 61 and 63 days of data were logged at Te Pahi, Oturori and Frenchmans, but only 31 at Pigs Gully. Generally, there was very low detection of bottlenose dolphins across the four locations, with the highest mean detection positive minutes per day of 0.72 and highest percentage of days with detections of 16% (Figure 3-15). The proportion of recording days with dolphin detections and mean DPM was substantially higher at locations outside the marine mammal safe zones – with the Te Pahi site having the highest relative presence of dolphins. There were no detections of dolphins throughout the monitored period at the Frenchmans site within the dolphin safe zone (Figure 3-15).

The relative presence of vessels was distributed similarly to dolphin presence (Figure 3-15), with a high percentage (approximately 60%) of monitored days with detections of vessel echosounders at the two sites outside the marine mammal safe zones (Te Pahi, Pigs Gully). Forty-three percent of monitored days at Oturori had detections of vessel echosounders, and only around 2% of days at Frenchmans. The Te Pahi site had the highest mean detection positive minutes for vessels (17 DPM/day) followed by Pigs Gully (Figure 3-15).

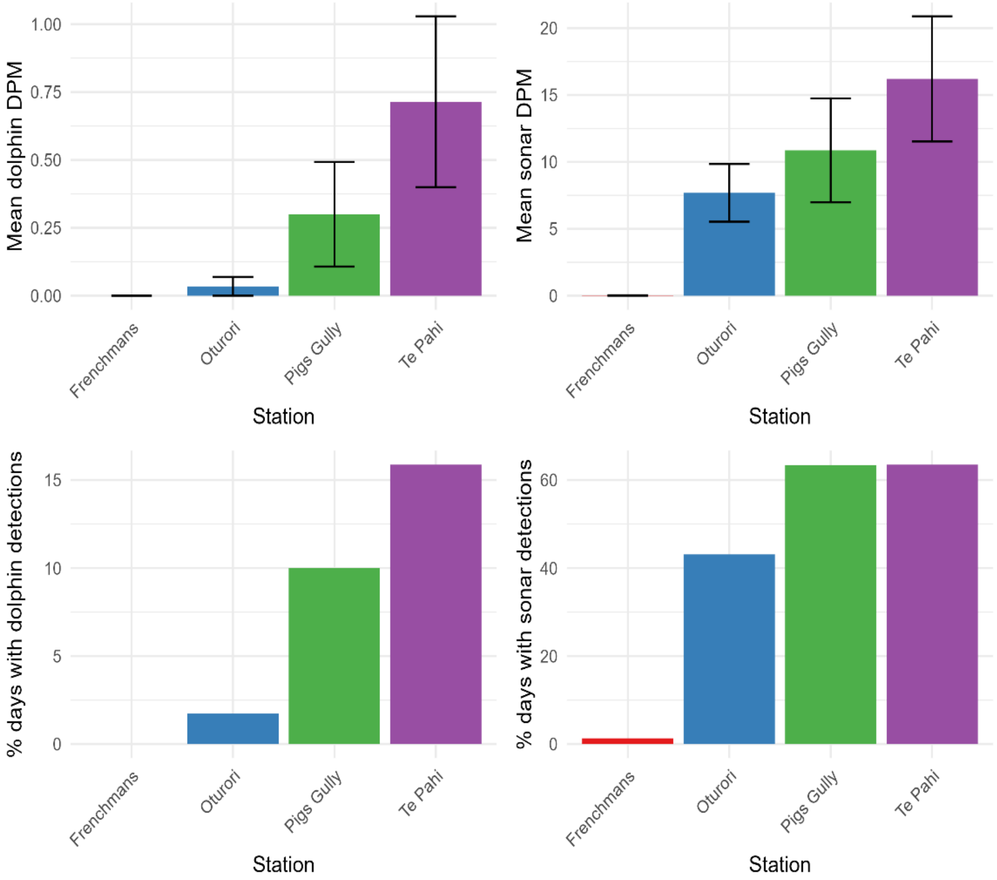

Figure 3-15: Acoustic monitoring results. Summary of metrics from the analysis of CPOD data for the relative occurrence of bottlenose dolphins (left) and vessels (right). Metrics for mean detection positive minutes (DPM) per day provided alongside the percentage of monitored days with detections. Site inside marine mammal safe zones are given on the left (Frenchmans, Oturori), outside safe zones are on the right (Pigs Gully, Te Pahi).

3.7 Vessel compliance and interactions

During the 17 dolphin encounters, we recorded 69 interactions (i.e., presence of dolphin groups within 300m) with boats. Twelve interactions occurred with permitted vessels and 57 with non-permitted vessels (Figure 3-16). Permitted vessels were compliant with the rules around the BOIMMS 100% of the time. However, non-permitted commercial vessels did not follow the BOIMMS rules on multiple occasions, bringing down the overall average rate of compliance for commercial vessels (Figure 3-16). The majority (>80%) of recreational vessels did not follow the regulations and there was limited difference between sail and power-driven vessels (Figure 3-16). We observed no breaches of rules preventing swimming with groups of marine mammals. Thus, all vessels (both recreational and commercial) were 100% compliant with this component of the BOIMMS rules.

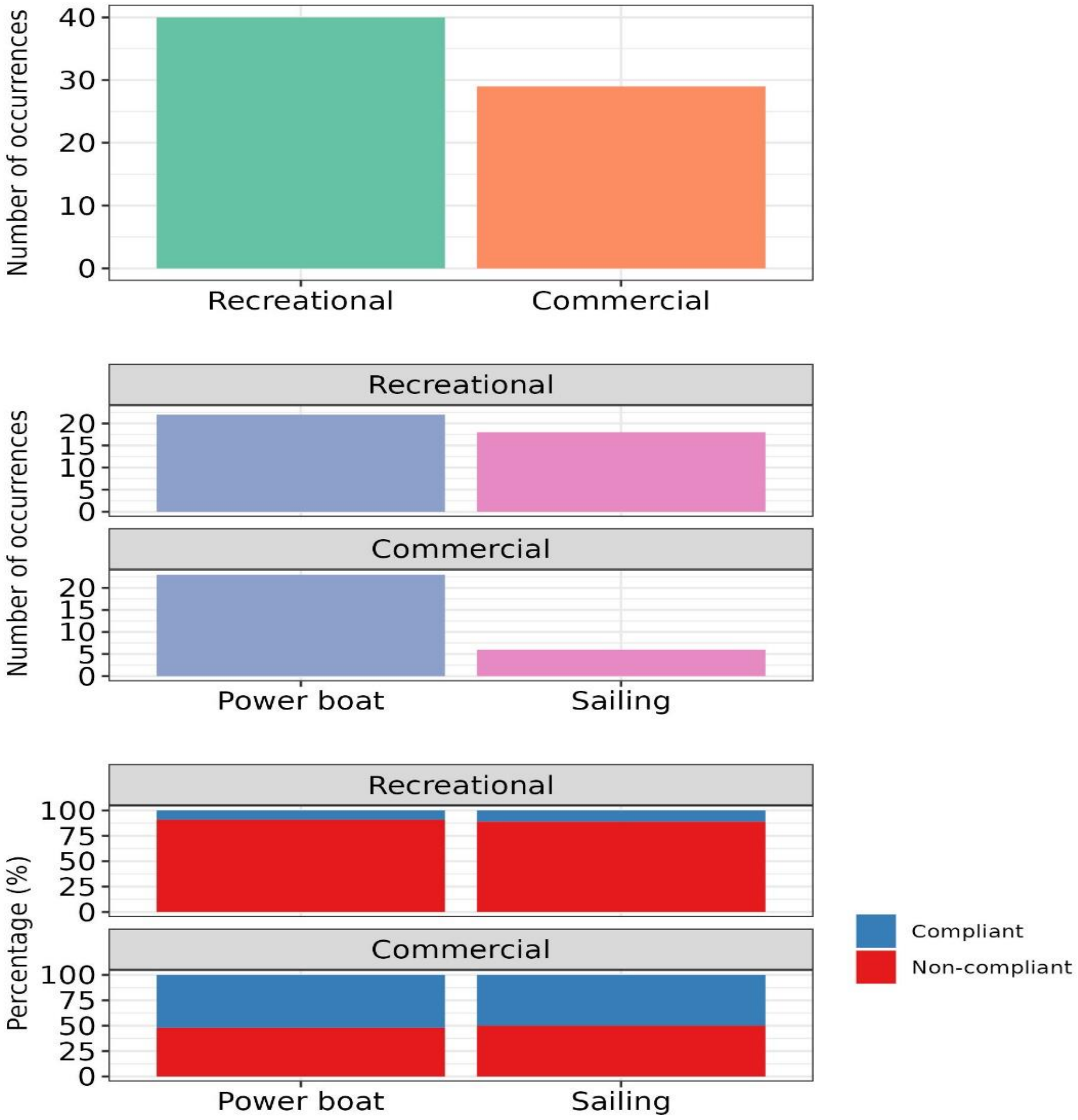

Figure 3-16: Vessel interaction and compliance. Vessel interaction and compliance. Summary of number of occurrences and percentage of compliance to the Bay of Islands Marine Mammal Sanctuary rules during dolphin encounters, during April/May 2024. Top plot summarises the number of dolphin interactions with

recreational and commercial vessels (bar plot), then these numbers are broken down by type of vessel in the middle plot (bar plot), and percentage of compliance by type of vessel in the bottom plot (stacked bar plot).

We carried out 24 compliance checks within the marine mammal safe zones (12 in each zone), over 13 days. All vessels observed crossing the safe zones were non-permitted. In total, 91 vessels were observed within these zones and were thus checked for compliance—eleven commercial and 80 recreational vessels. Irrespective of commercial or recreational categorisation, more than 50% of the vessels did not comply with the 5-knot speed limit within the zones (Figure 3-17). Non-powered vessels had a higher rate of compliance to the 5-knot rule, likely due to the speed limitations of sailing in enclosed waters. The majority (>75%) of power-driven vessel of both recreational and commercial classes did not comply with the 5-knot speed restriction in the marine mammal safe zones.

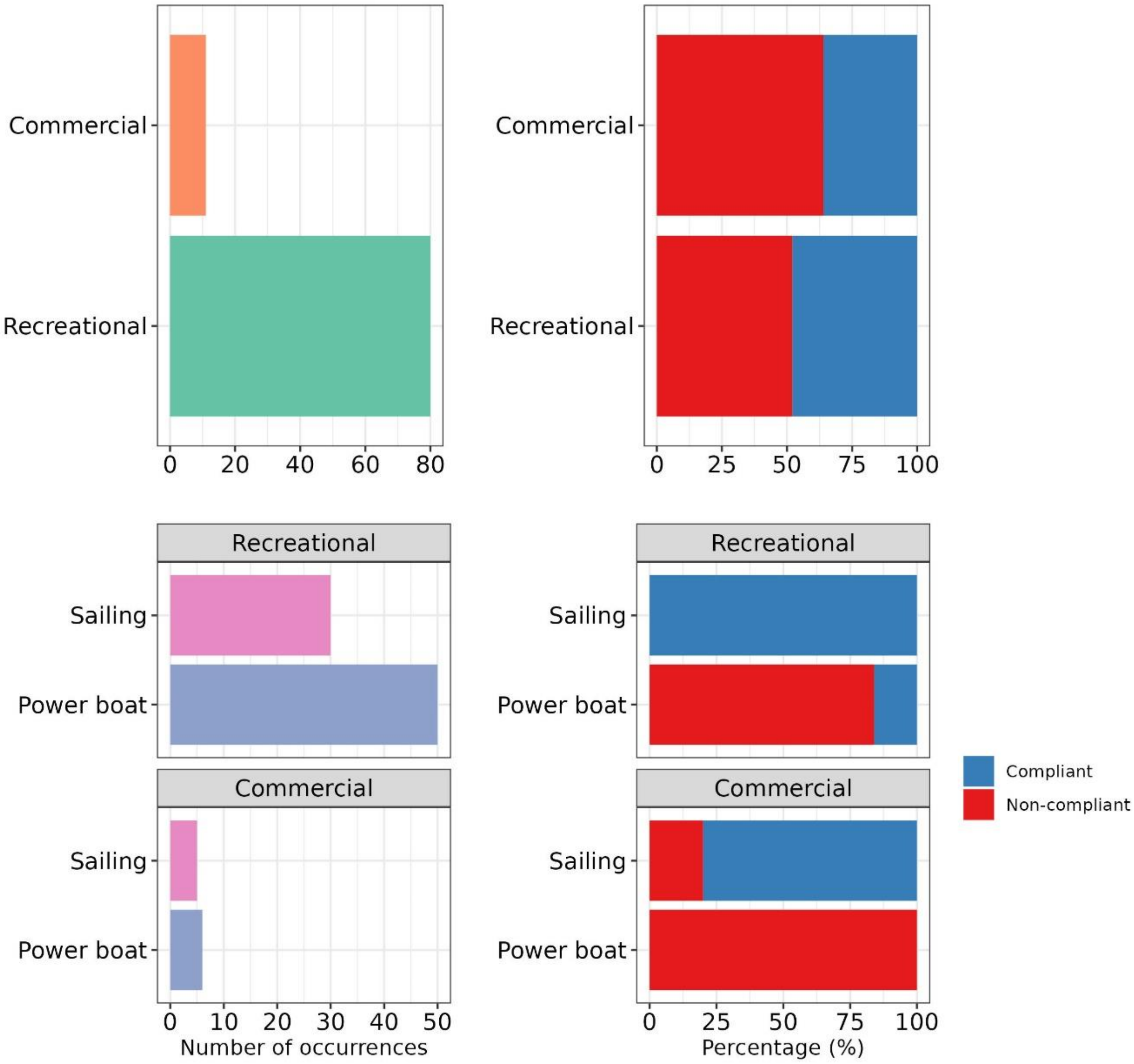

Figure 3-17: Dolphin safe zone compliance. Summary of number of occurrences and percentage of compliance to the Bay of Islands Marine Mammal Sanctuary rules within the designated safe zones, during April/May 2024. Plots in the top row summarises the number of dolphin interactions (left) and percentage of compliance (right) from recreational or commercial vessels, and plots in the bottom row break those numbers down by type of vessels.

4 Discussion

This study has provided several key outputs that will aid in the management of bottlenose dolphins in the Bay of Islands. Most notably, it integrates population monitoring data from 1994 to 2024. A dataset of this duration, with short temporal gaps, is a significant asset for managing populations of long-lived, slowly reproducing species and provides DOC and its partners with the ability to explore a variety of questions related to the local population's previously reported decline. Along with this report, three curated datasets are submitted to DOC: 1) a comprehensive photo-identification database containing capture histories of all individuals encountered in the Bay of Islands between 1994 and 2024; 2) an encounter database detailing observations of bottlenose dolphins groups during research over the same period; and 3) a calf mortality database documenting outcomes of calves born in the Bay of Islands with confirmed mother ID. Additional data on dolphin behavioural responses to vessels and acoustic data collected during this study will also be made available to DOC within twelve months of the date of this report.

4.1 Caveats and limitations

While all efforts were taken to standardise data across multiple institutions and researchers, it is inevitable there will be some inconsistencies over a database of such long duration which results in some important limitations. These inconsistencies include variability in the ability of individual researchers to recognize and document associations among individuals (e.g., mother – calf associations), variation in the quantity of research effort and variability in the quality of photo-identification images among researchers involved in this programme. While approaches to collect data on calf mortality have been largely standardised over the study duration (i.e., confirmation of mother identity over multiple, independent encounters), it was out of scope of this project to review photographic documentation of such associations and thus we assume each mother identification is robust. While data on calf mortality has been collected throughout the study period, it was not always a key research question for individual researchers and thus there may be some discrepancies in the amount of effort invested into confirming mother identity. Incorrect documentation of mother identify will result in bias estimates of calf mortality and reproductive rate and thus any differences between periods (e.g., the recent decrease in apparent first year mortality) may be due to variation in the robustness of the assignment of mother identity to individual calves.

Variation in the amount of research effort across the duration of the study may also impact the estimation of demographic parameters. Calf mortality and reproductive rate in particular are likely to be biased by such variation. Unlike local abundance and apparent survival estimation, reproductive parameters are not directly modelled with mark-recapture approaches and thus cannot incorporate variation in the probability of capture/recapture, which can reflect variation in research effort. For example, the number of reproductive females seen each year is clearly a product of how much effort has been utilised to survey the local population. Similarly, in years with higher effort there is greater ability to confirm the occurrence of calves and their mothers' identity, and to determine the fate of calves. Thus, years with high effort will have more accurate estimates of both calf mortality and reproductive rate. Many years that exhibit low calf mortality rates (e.g., 2000 to 2003 and 2017 to 2021) have low effort and the calculation of mortality is based on the fate of very few known calves (in most years a single calf). If effort were higher, there would be greater chances to observe calves that do not survive, thus low effort can underestimate mortality. Estimates of apparent calf mortality for years with low effort should be treated with caution.

Bias associated with variation in the quality of photo-identification images used for analysis was minimised in this study by the use of strict controls on mark distinctiveness and photo quality which resulted in the removal of individuals from the previously used photo-identification catalogues where image quality was not as strict as that required for the calculation of demographic parameters (section 2.2). These interventions are important to uphold critical mark-recapture assumptions around the loss of marks and mark change over time. It was outside of the scope of this project to review individual matches to the catalogues by all previous researchers and thus it is not possible to account for possible discrepancies in the interpretation of photo quality and distinctiveness categories (all researchers used the same scoring procedure).

Several assumptions of the mark-recapture approaches we used to estimate local population abundance (and other parameters) were not upheld (Appendix A), and we followed standard interventions to account for variability in capture probabilities, transience and overdispersal in the binomial capture-histories. In addition, we tested the assumption of closure within the primary periods of the robust design analysis, which confirmed closure all but eight of the forty-seven primary periods (Appendix A). Some studies that use seasonal primary periods to estimate population parameters in bottlenose dolphins infer population closure due to the comparatively short duration of a single season relative to the life-span (up to 50 years) of an individual and thus low chances of recruitment/permanent removal from the population (Fruet et al., 2015, Methion & Diaz Lopez 2018). It is likely, therefore, that the lack of closure in some primary periods in this study is due to our observed high rates of and variability in temporary emigration (Kendall 1999). Further, biases introduced by a lack of closure are typically increased in cases where populations exhibit Markovian emigration, impacting accuracy and precision of parameter estimates (Kendall 1999, Hamilton 2013). Thus, estimation of local abundance for the eight primary periods where closure could not be confirmed should be treated with caution. These periods occurred sporadically throughout the study, and thus any biases in the estimates do not influence the overlying trends in local abundance.

A further assumption of mark-recapture analysis is the opportunity for meaningful mixing of individuals between capture periods (Lukacs 2009). For the local population 'snapshot', there was a period of 5 days between capture and recapture period and there was a minimum of 7 days between primary periods for the robust design (median = 29). Given the rates of transience in this local population, 5 days may not be adequate for appropriate mixing between capture periods. Yet, given the short duration of sampling and the low encounter rate observed during fieldwork in this study, a longer period between capture and recapture periods was not possible. However, the close agreement between the 'snapshot' of local abundance and that produced using the robust design for the same period suggest any impact from a lack of mixing is minimal.

4.2 Population demographic analysis

The long-term analysis of demographic parameters for bottlenose dolphins in the Bay of Islands confirmed several periods of decline and vulnerability of this local population (Figure 3-6). Our analysis, using mark-recapture approaches which are comparable to previous demographic assessments in the Bay of Islands, reveal the local population has continued to decline in abundance. The lowest point of abundance occurred at summer 2019 with 16 individual dolphins (Figure 3-6). The highest summer estimate occurred in 1997 with 244 individuals. Thus, even while considering seasonal fluctuations in abundance, the number of dolphins using the Bay of Islands has experienced a substantial decline, greater than that which was previously documented (Tezanos-Pinto et al. 2013). The estimates of abundance generated during this study, using the long-term photo-identification database, have both similarities and differences with those generated by previous studies on this local population (Hamilton 2013; Tezanos-Pinto et al. 2013). Using the robust design, Tezanos-Pinto et al. (2013) found that total local abundance of adult dolphins peaked at 240 (95%CI = 99–581) during a primary period in October 1997 which is similar to our estimate of 207 (95%CI = 140–306) for spring 1997. The study conducted by Tezanos-Pinto et al. (2013) estimated a low in local abundance at September 2005 (of 21 individuals) that varied considerably from our estimate of 109 individuals for spring 2005 (see next paragraph for further explanation of this difference). However, the seasonal (POPAN) models of Tezanos-Pinto et al. (2013) for spring 2004 and 2005 agree closely with the estimates presented in this study (approximately 115 individuals). Estimates of monthly abundance by Hamilton (2013) for 2009 are broadly similar to the results presented here; however, Hamilton (2013) also provided estimates for 2012 which were not included here due to a lack of data meeting our requirements for secondary periods.

It should be noted that while the two previous studies of local population abundance in the Bay of Islands used the same data and the same mark-recapture approach as that used in this study, some differences in the study design should be considered when comparing results. For example, the RD analysis of Tezanos-Pinto et al. (2013) and Hamilton (2013) pooled primary periods into discrete blocks that roughly aligned with seasons but were shorter in duration (approximately 1 week long), with individual days as secondary periods. Such primary periods were configured to better account for fine-scale temporal patterns of habitat use in the study area (i.e., discern rates of movement in and out of the Bay of Islands) (Tezanos-Pinto et al. 2013). In this study, our aim was to generate robust demographic parameters that utilise as much of the available capture history data as possible, minimising gaps in the time series and delivering estimates at temporal scales that are useful for management. Thus, we pooled our primary periods at broader, seasonal scales which is consistent with the majority of studies using RD to estimate population parameters in bottlenose dolphins elsewhere (Silva et al. 2009; Smith et al. 2013; Fruet et al. 2015; Santostasi et al. 2016; Methion & López 2018; Durden et al. 2021). The POPAN analyses used by Tezanos-Pinto et al. (2013) pooled data at seasonal scales and is thus most comparable to the results of this study. However, the different parameterisation of the POPAN model (Schwarz & Arnason 2009), especially the lack of inclusion of temporary emigration, should be considered when comparing the results of these two studies. Additionally, variation in the way the transience and capture heterogeneity were accounted may influence the comparability of results between studies.

In analysis of long-term photo-identification datasets, variation in demographic parameters between studies is expected, as the longer the dataset, the more information is available concerning the fate of individuals (Pollock 1991). While previous studies delivered results that were well-suited for their application, the results presented in this study are likely more insightful for the characterisation of long-term variability in population demographics. Despite these considerations, the broad similarity of the estimates for local abundance and other population parameters between this study and previous research confirms the trends independently exhibited by the three studies. In terms of survival estimates, similar to this study both Tezanos-Pinto et al. (2013) and Hamilton (2013) found RD models with constant survival fit the observed capture history better than time-varying parameterisations. Estimates of apparent survival by Tezanos-Pinto et al. 2013 for the period 1997 to 2006 were 0.928 (SE = 0.008) using the RD and 0.85 (SE = 0.019) using POPAN and excluding transients which varied slightly from those presented in this study for resident individuals (0.90). In contrast, Hamilton (2013) estimated survival at 0.63 (SE 0.05, 95% CI = 0.53-0.72), considerably lower than that presented here. It is likely that these differences are due to the way in which transient individuals are accounted for. Hamilton (2013) included no parameterisation of transience within the

RD models, which likely biases these estimates low (as mortality is confounded by individuals only seen once). Tezanos-Pinto et al. (2013) removed the first capture event for survival rate estimation, which is an effective approach to remove transient individuals (i.e., those only seen once), but may remove some instances of true mortality.

Temporary emigration was estimated to increase over the study period (Figure 3-5), with an increase in the probability of remaining outside of the study area (γ′) over time in the later time period (2014–2024) (Figure 3-5). The probability of moving outside of the study area (γ′′) remained constant over the study period at 0.143 (95%CI = 0.123–0.165). Tezanos-Pinto et al. (2013) reported high variability and a lack of precision around rates of temporary emigration with a full-time varying specification. However, that study found that temporary emigration was random rather than Markovian. Hamilton (2013) showed that patterns of emigration changed to Markovian, a result which was consistent with this study. It has been suggested that the change in temporary emigration patterns is due to variation in the way the wider north-eastern bottlenose population uses the Bay of Islands, with the area being utilised by a smaller number of frequent users, with less visitation by occasional visitors and transients (Hamilton 2013) (i.e., fewer non-resident individuals using the area). The increase in the probability of remaining outside the study area (γ′) seen in this study supports this hypothesis and provides some context for the long-term decline in local abundance. The increase in temporary emigration coincides with the most recent period of dramatic decline in abundance (2014–2015). There is no evidence for increased mortality in the Bay of Islands judging by the frequency of reports of carcasses and thus emigration is likely the predominant cause of declines in the number of dolphins utilising the area.

Calf mortality and reproductive rates have continued to be highly variable in the Bay of Islands, with calf mortality being particularly high compared to other populations (see Tezanos-Pinto et al. 2015 and reference therein). It should be noted, however, that comparison to calf survival estimates calculated for the population in Doubtful Sound (Currey et al. 2009) are not valid, as the method of calculating survival rates is different from the mortality rates calculated here. However, viewing rates of mortality by breeding season helps to provide context around the potential relationships between population parameters (i.e., survival and abundance) and may allow for more causal links to be investigated between potential stressors and population health (Currey et al. 2011; Meyer et al. 2015).

Significant variability in bottlenose dolphin population abundance and other demographic parameters has been reported elsewhere in New Zealand (Currey et al. 2007; Currey et al. 2009) and internationally (Bejder et al. 2006; Coscarella et al. 2012; Van Cise et al. 2021). In Fiordland, historical decline in the abundance of the bottlenose dolphin population of Doubtful Sound has now stabilised, with the population increasing to levels seen historically (Bennington et al. 2020). Similarly, the population in Dusky Sound has shown several periods of decline and recovery (Bennington et al. 2020). Stochasticity in population vital rates is expected in small populations of marine mammals. However, the multiple periods of decline and the lack of recovery in local abundance in the Bay of Islands over a thirty-year period highlights the significant issues faced by this local population.

As discussed in previous studies on this local population, the most likely scenario for the decline in local abundance in the Bay of Islands is a decrease in the frequent users of the Bay of Islands and a reduction in both occasional visitors and transient dolphins (Hamilton 2013; Tezanos-Pinto et al. 2013; Peters & Stockin 2016; Guerin 2022). The results of this study support this theory, and suggests that the rates of emigration (i.e., decline in the use of the Bay of Islands) have increased. Individuals in the Bay of Islands form part of the wide-ranging north-eastern NZ coastal bottlenose population, with individuals sighted in the Bay of Islands also being sighted within the Hauraki Gulf (Berghan et al. 2008) and ranging as far as the Manukau Harbour on the West Coast (Constantine 2002). Recent population monitoring work in Bream Bay has documented the presence of a large local population of bottlenose dolphins (288 individuals; Brough et al. 2024). At least 37 individuals (6%) overlap between the Bay of Islands and Bream Bay catalogues, and the two areas are relatively close (~100 km apart). Thus, other areas in Northland may be becoming preferred habitat over the Bay of Islands. Despite research documenting likely stressors in the Bay of Islands such as the impacts of vessels and tourism (Constantine et al. 2004; Peters & Stockin 2016), the mechanism for the decline remains largely unknown. With the long-term dataset now available, additional research is required to determine potential causal relationships between variability in population parameters and potential stressors. The most recent decline in local abundance (as of 2015) does not coincide with a change between researchers, or any significant change in the amount of effort (although there is a change in both researchers and a decrease in effort from 2016). Further, comparatively low abundance was observed by our field surveys which had high effort compared to the 2016 to 2023 period. Thus, it is unlikely the most recent decline can be explained by variation in survey design or photo-identification techniques. Potential reasons for the rapid decline between summer and autumn 2015 should be investigated further and may reveal possible mechanisms related to wider observed decline in this local population.

4.3 Spatiotemporal distribution

By pooling data over three decades and using a SDM approach, the analysis of spatial and temporal distribution generated in this study provides the best available information on habitat use for this local population. Our analysis revealed a strong seasonal preference in both the occurrence and distribution of dolphins within the Bay of Islands. Such seasonal patterns have been reported previously (Constantine & Baker 1997; Hartel et al. 2014), and align closely with the results observed in this study. Further, the lower rates of habitat suitability during the cooler months of the year support our finding of lower seasonal abundance during this time (Figure 3-7). Seasonal patterns in distribution are the norm for many populations of coastal delphinids, and have been exhibited in multiple locations around New Zealand (Henderson 2013; Brough et al. 2019; Dwyer et al. 2020; Bennington et al. 2023). Thus, it is not surprising that our study confirmed seasonal patterns in the Bay of Islands. However, confirming the existence of such patterns and the locations of relative seasonal importance provides options for temporally dynamic management.

Interestingly, our study found that the areas of importance within the Bay of Islands were largely consistent over time (Figure 3-11). Hartel et al. (2014) found that broad-scale distribution patterns within the Bay of Islands had also remained consistent over a 10-year period, but that fine scale patterns had changed considerably. The areas of importance (i.e., hotspots) in distribution estimated by Hartel et al. (2014) for 2007-2010 and by Peters & Stockin (2016) were highly consistent with those identified in the latter two periods of this study. Our results suggest the major hotspots in distribution (e.g., between Tapeka Point, the Black Rocks and Motuarohia Island) are consistent over the majority of seasons and over multiple decades. The stability of hotspots in distribution have also been exhibited in bottlenose dolphins in Dusky Sound (Bennington et al. 2023) and for Hector's dolphins (*Cephalorhynchus hectori*) at Banks Peninsula (Brough et al. 2019), despite variability in population abundance in both locations (Gormley et al. 2005; Bennington et al. 2020).

In general, the Bay of Islands study area has increased in relative habitat suitability over time (Figure 3-11). This finding suggests that the environmental characteristics of good quality habitat have increased and provides useful insights for the declines in the number of dolphins using the area. The lack of congruence between the trends exhibited by the spatial models and the results of the population modelling could be due to several factors including the lack of inclusion of abundance data in the SDMs or the stressors on the population. Of our available dynamic variables, the SDMs revealed positive associations between occurrence and low turbidity, low chlorophyll-*a* concentration and high sea surface temperature. Associations between low turbidity and occurrence/abundance have been observed in other populations of coastal dolphin in New Zealand (Brough et al. 2023; Bennington et al. 2024) and may be related to foraging success (Brough et al. 2020). Turbidity in coastal environments is a major stressor that can have broad consequences for benthic and pelagic communities (Brown et al. 2017; Morrison et al. 2023). Chlorophyll-*a* concentration is a proxy for phytoplankton productivity, however the relationship with highly mobile predators is often complex and scale dependent (Mannocci et al., 2017). The lack of a negative correlation between SST and habitat suitability is interesting in light of known increases in mean SST, the prevalence of marine heatwaves in this region and their broad impacts on habitat quality (Shears & Bowen 2017; Montie et al. 2024). However, further refinement of the SDMs used here is required before they can be used to investigate the potential impact of stressors (Brough et al. in review).

Predicted patterns of occurrence based on habitat suitability index provide important information on the location of core areas (i.e., hotspots) across different spatial and temporal scales. However, it is important to note the distinction between occurrence and abundance, where different environmental factors may influence the spatial abundance of species resulting in distinct patterns over spatiotemporal scales (Jiménez-Valverde et al. 2021). The lack of alignment between our results exhibiting the periodic declines in abundance, and increasing habitat suitability may be due to the lack of incorporation of abundance data (i.e., group size) into our SDMs. For example, the frequency of sightings (i.e., occurrences) of dolphins in the Bay of Islands may not have varied considerably over time, but there is some evidence for variability in group size (Hamilton 2013, Peter & Stockin 2016). Initial attempts to model abundance in this study were unsuccessful, largely due to an inability to meaningfully scale group size estimates by spatially explicit effort, an important consideration for abundance models (Waldock et al. 2022; Brough et al. 2023). Further research should work to refine the SDMs developed here which would enable more nuanced interpretation of variation in distribution, habitat use and potentially, the impact of stressors.

4.4 Compliance and effectiveness of the BOIMMS

The results of our monitoring of compliance to the BOIMMS rules show generally poor adherence to the rules around dolphin interactions and the safe zones across most categories of vessels. However, vessels were 100% compliant with rules preventing swimming with dolphin groups. For the rules around interaction with dolphins (i.e., coming to a full stop within 300m), only permitted commercial vessels regularly adhered to the rules. Non-permitted commercial vessels were mostly non-compliant to rules on dolphin interactions and the vast majority (>80%) of recreational vessels were non-compliant. These results suggest that there is either 1) a considerable lack of awareness of the rules among non-permitted commercial vessels and recreational vessels in particular, or 2) a lack of respect for the rules. For either scenario, both increased on-water and off-water education, compliance and enforcement of the rules are required to ensure the BOIMMS is effective in limiting impacts on bottlenose dolphins and other marine mammals. Further work should also investigate compliance with wider regulations under the Marine Mammal Protection Regulations (1992).

Ninety-one vessels were monitored in proximity to the two marine mammal safe zones during our field programme. Of these, the majority of vessels were non-compliant to the 5-knot speed limit (Figure 3-16). Of the vessels that were compliant, most were sail-driven vessels that were likely unable to exceed 5 knots due to lack of wind speed. Power-driven vessels were the most commonly non-compliant across both recreational and commercial categories. Similar to the rules around interactions with dolphins, the lack of compliance of vessels to the 5-knot rule in the marine mammal safe zones significantly limits the effectiveness of the BOIMMS. Both the impacts of noise pollution and vessel strike increase with the vessel speed (Constantine et al. 2015; Findlay et al. 2023), and thus power-driven vessels likely have a larger impact on the local population. If DOC and its partners wish to maintain the marine mammal safe zones, there needs to be considerably more attention given to education, compliance and enforcement, particularly of power-driven vessels, for these tools to be worthwhile.

Our analysis of long-term population demographic parameters, spatial distribution and habitat use also suggests the effectiveness of the BOIMMS for halting and reversing the decline in the local bottlenose dolphin population may be limited. While it may be too early to notice substantial changes in habitat use, there is no indication that more individuals are using the Bay of Islands following the implementation of the BOIMMS (i.e., no meaningful increase in local abundance). Our snapshot of spatial distribution patterns for Autumn 2024 (Figure 3-14) and the results of the acoustic monitoring (Figure 3-15) suggest that the marine mammal safe zones are not regularly used by the dolphins, at least during the time of year where data was collected under this study. In particular, that the acoustic monitoring in the dolphin safe zone in Frenchmans made no detections of dolphins over several months calls into question the relevance of this safe zone. Further, the distribution modelling using the long-term data suggest the safe zones do not generally coincide with the key hotspots of distribution. While habitat suitability in the safe zones is highest during the summer months, this is related to the general increase in habitat suitability across the BOIMMS during summer, rather than these areas having high relative importance. Positioning fine-scale protection areas is typically better undertaken in a marine spatial planning (MSP) approach where data on the distribution of biodiversity (i.e., the dolphins) and potentially stressors (i.e., vessels) is systematically analysed along with input from stakeholders (Pınarbaşı et al. 2017). With insights from the dynamic SDMs developed in this study, an MSP approach could incorporate variation in seasonal distribution to establish dynamic spatial tools that overlap key areas for the dolphins at various times of the year. Such an approach is particularly relevant for this local population given challenges with the effectiveness of static protection (Hartel et al., 2014; Guerin 2022). Dynamic tools often allow the refinement of areas that are more meaningful for biodiversity, and have better uptake and compliance due to the integration of stakeholder concerns (Collie et al. 2013).

While our fieldwork programme was limited in duration for a full, cross-season investigation of compliance with the rules of the BOIMMS, our results provide useful insights as to the degree of compliance during a popular time of the year (e.g., easter, school holidays, ANZAC weekend). Further work should undertake an assessment of compliance over a full year, especially during the summer period, where interactions between vessels and dolphins and vessel traffic is highest (Peters & Stockin 2016, Guerin 2022). Additionally, understanding any change in the rate of interaction between vessels and marine mammals and any associated impacts on behaviour, is a critical part of understanding the effectiveness of the BOIMMS which requires further investigation.

4.5 Conclusions and ongoing research

The research undertaken in this study provides DOC and its partners with the best available information for the management of bottlenose dolphins in the Bay of Islands. The study has provided clear evidence for multiple periods of decline with no indications of recovery to levels of abundance seen during the first decade of research in this area. While recognising the limited duration (20 days across Mar – June 2024) of compliance data collected by this study, the effectiveness of the BOIMMS appears limited by a lack of compliance, particularly with the marine mammal safe zones. Additional investigation of compliance and the efficacy of the sanctuary at reducing vessel interactions (and associated impacts on behaviour) is required to establish a thorough representation of the effectiveness of the BOIMMS and the need to refine management to protect this important local population.

Opportunities for additional research to support the BOIMMS are also apparent, particularly with the integration of long-term datasets on this local population. Such research could include:

- **Potential Consequences of Disturbance (PCoD) modelling:** Despite some evidence on the impacts of vessels on dolphin behaviour, there is limited information to link this with the decline in local population abundance. Several modelling approaches (including PCoD) can be used to determine potential mechanisms behind variability in demographic parameters, including multiple stressors and determination of the likely population vital rates (e.g., migration, calf survival) behind the decline in abundance. Such modelling is routinely carried out in the management of protected species at the national level and has several local scale examples in New Zealand (e.g., Currey et al. 2011).
- **Ecological factors:** Investigation of ecological factors behind the variability in local population abundance should be considered, particularly given the perilous state of a range of marine species and habitats in the Bay of Islands. Basic information on prey preference and availability, prey habitat selection, variability in foraging success and the relationship between these factors and changing conditions is vital for ecosystem-based management of bottlenose dolphins in the Bay of Islands.
- **Genetic assessment:** the genetic diversity and isolation of bottlenose dolphins in the Bay of Islands has been evaluated, however more research is needed to understand the sociogenetic structure of the local population, relatedness among individuals and its relatedness to nearby local communities.
- **Marine spatial planning:** Spatially explicit information on dolphin occurrence and the distribution of potential stressors (e.g., vessels of different categories, marine heatwaves) is available which would enable a systematic spatial planning analysis to identify scenarios for spatial based management of this local population, guided by stakeholder input. NIWA has recently developed guidelines for such processes for DOC (e.g., Lundquist et al. 2021) and regularly uses these approaches throughout New Zealand.
- **Ongoing monitoring**: Above all, it is critically important to continue the population monitoring of bottlenose dolphins in the Bay of Islands to determine if/when management interventions are effective.
- An integrated approach to monitoring the wider north-eastern population in key hotspots (e.g., Bay of Islands, Bream Bay, Hauraki Gulf), should also be considered to place local declines within the context of the wider population health.

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Appendix A Mark recapture assumption testing

Table A-1: RD assumption tests. Testing of assumptions for the robust design models in R2UCare. Test2 functions (Test.CL and Test.CT) test for heterogeneity in probability of capture and Test3 tests for transience. A p-value <0.05 indicates the failure to uphold an assumption and merits the interventions undertaken in our study design.

Test	Statistic	Degrees of freedom	p-value
Test 2.CL	582.495	44	<0.000
Test 2.CT	186.396	62	<0.000
Test 3.SR	60.232	34	0.004

Table A-2: Closure tests. Testing for closure within primary periods of the robust design. A p-value <0.05 indicates the assumption of population closure is not upheld, shaded here in orange.

Period	statistic	p-value
Su1997	3.739	1.000
Au1997	6.634	1.000
Wi1997	5.013	1.000
Sp1997	-4.310	0.000
Su1998	7.045	1.000
Au1998	-0.621	0.267
Wi1998	1.636	0.949
Sp1998	4.253	1.000
Su1999	-0.069	0.473
Au1999	-2.641	0.004
Wi1999	4.817	1.000
Sp1999	2.934	0.998
Su2000	-2.165	0.015
Wi2003	0.792	0.786
Sp2003	3.153	0.999
Su2004	4.904	1.000
Wi2004	-0.515	0.303
Sp2004	6.209	1.000
Su2005	0.773	0.780
Wi2005	0.189	0.575
Sp2005	3.425	1.000
Au2006	-2.828	0.002
Wi2009	5.940	1.000
Sp2009	8.880	1.000

Period	statistic	p-value
Su2010	-1.692	0.006
Su2013	-3.640	0.000
Au2013	0.035	0.514
Wi2013	-0.566	0.286
Sp2013	3.311	1.000
Su2014	0.783	0.783
Au2014	-1.199	0.115
Wi2014	1.817	0.965
Sp2014	5.668	1.000
Su2015	3.027	0.999
Au2015	-0.424	0.336
Sp2017	2.530	0.994
Su2018	-1.328	0.092
Su2019	1.789	0.963
Wi2019	1.060	0.855
Sp2019	1.673	0.953
Su2020	-0.136	0.446
Su2021	-1.767	0.039
Au2021	-0.537	0.296
Sp2021	-2.000	0.023
Au2022	-2.121	0.017
Wi2022	1.414	0.921
Au2024	4.971	1.000

Appendix B Predicted HSI uncertainty

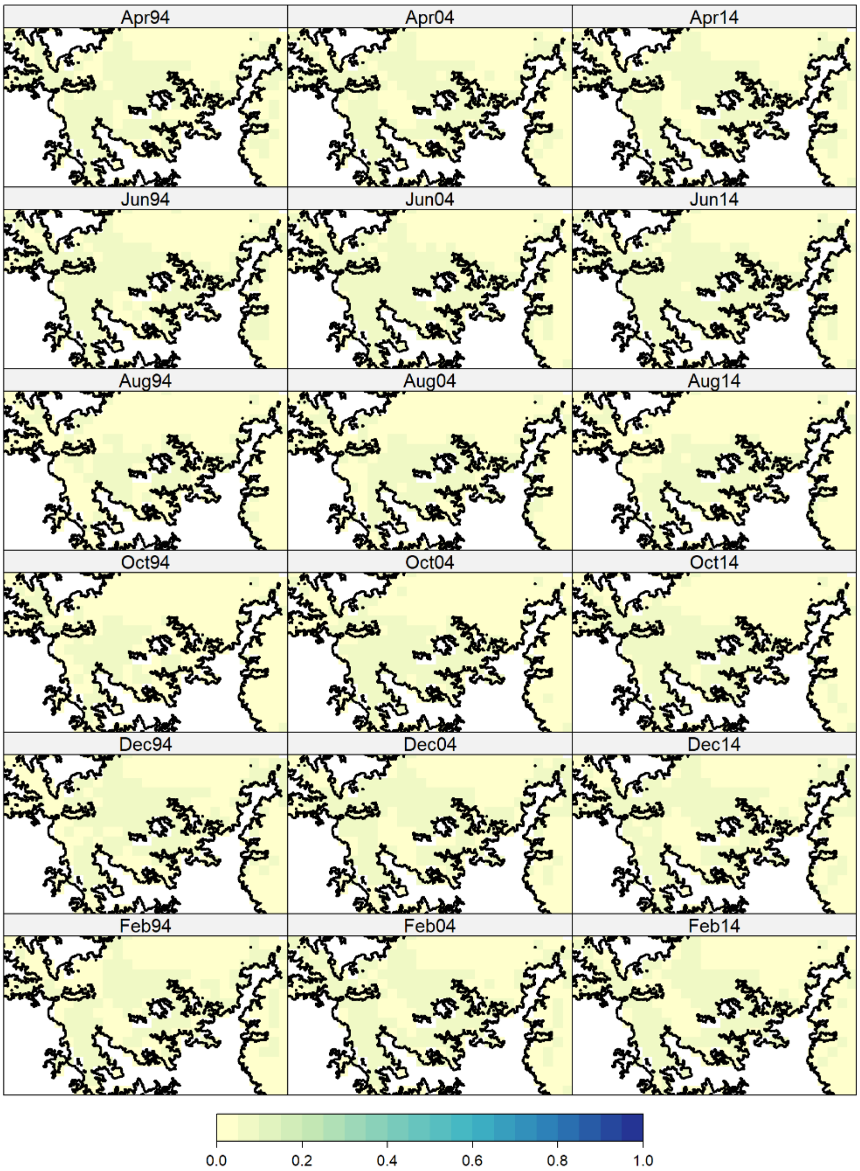

Figure B-1: Uncertainty in predicted distribution of coastal bottlenose dolphin.redicted uncertainty (SD) around mean habitat suitability index for bottlenose dolphins (0-1 indicated by colour key) from species distribution modelling across the study area for three time periods (columns left to right: 1994–2003, 2004–2013, 2014–2024), for six months (rows top to bottom: April, June, August, October, December, February).