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Long- and short-term impacts of vessels on Hector's dolphins (*Cephalorhynchus hectori*) at Te Pātaka-o-Rākaihautū/ Banks Peninsula

William J. Rayment, Steph Bennington, William Carome, Peter Dillingham,
Elisabeth Slooten, Lindsay Wickman and Stephen M. Dawson



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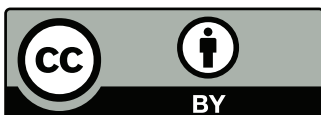
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Long- and short-term impacts of vessels on Hector's dolphins (*Cephalorhynchus hectori*) at Te Pātaka-o-Rākaihautū/Banks Peninsula

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Abstract

Vessel activity has a variety of potential impacts on cetaceans. Hector's dolphin (*Cephalorhynchus hectori*) is particularly vulnerable to these impacts at Te Pātaka-o-Rākaihautū/Banks Peninsula, where there is a thriving tourism industry. We used photographic and dolphin distribution data gathered between 1984 and 2022, supplemented with contemporary distribution data, to estimate demographic parameters of Hector's dolphins and the effects of vessel activity. Mean apparent survival rate in 2009–2019, following protection from harmful fishing methods, was higher than before protection (1986–1989) in 72.3% of model runs. The proportion of calves in the population varied from 0.007 in 2020 to 0.084 in 1995, with no evidence of a linear trend over time. We estimated abundance of Hector's dolphins using the area between Birdlings Flat and Steep Head in 2020 at 1,868 (95% HDI 988–3,154). Analysis of sighting data from Akaroa Harbour indicated a change in dolphin distribution following a rapid expansion in cruise ship tourism in 2012. The core habitat shifted towards the outer harbour, and moved away from the region encompassing the two primary anchorages for cruise ships. Acoustic monitoring was used to measure the occurrence of dolphins in the middle harbour in 2019–2020. Detections decreased in relation to: increasing traffic from motor vessels, the presence of cruise ships and high levels of dolphin tourism. The demographic results suggest that the Hector's dolphin population at Banks Peninsula is now approximately stable. However, given the evidence that Hector's dolphins are displaced from core habitat by vessel traffic, we suggest immediate management to reduce the vessel pressure on dolphins in Akaroa Harbour.

Keywords: cetacean, cetacean watching, vessel impacts, ecotourism, dolphin, habitat use, human disturbance, threatened species

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1. Introduction

Ships, boats and other marine vessels can pose serious risks to cetaceans (whales, dolphins and porpoises). Collisions with hulls or propellers of vessels can cause injury or death. Motorised vessels can be a major source of noise pollution that may displace cetaceans from their preferred habitat. The presence of vessels may change cetacean behaviour in ways that disrupt behaviours critical for their survival (Rolland et al. 2012). Indirectly, the anchoring and manoeuvring activities of vessels may degrade habitats that cetaceans rely on (Broad et al. 2020).

These risks can be made worse by vessels actively seeking interactions with cetaceans for tourism.

1.1 Growth in cetacean watching and cruise ship tourism

Commercial cetacean watching began in 1955 in the USA (Hoyt 1995). This ballooned into a global industry worth billions of dollars per year (O'Connor et al. 2009), with the potential to continue growing rapidly (Cisneros-Montemayor et al. 2010; Mallard 2019).

Cetacean watching is often described as a form of ecotourism (Bejder and Samuels 2003). It creates opportunities for education, and may prompt customers to engage in conservation action or to donate to conservation charities (Orams 1997; Jacobs and Harms 2014). However, a growing body of evidence shows that cetacean tourism is generally not benign, due to the increased risks of boat strike (Wells and Scott 1997) and the physiological stress caused by vessel noise (Rolland et al. 2012). Other effects can include cetaceans expending more metabolic energy (Christiansen et al. 2014) and the disruption of key behaviours such as foraging and reproduction (Lusseau et al. 2009; Christiansen et al. 2010).

Cruise ship tourism has increased at the same time as the growth of cetacean tourism. The number of passengers embarking on cruise ships worldwide grew by 5.7% annually between 2008 and 2018 (CLIA 2019). In Aotearoa New Zealand, cruise ship passengers increased by 24% between 2018 and 2019 (Stats NZ 2019).

The impacts of cruise ships on marine species are largely unstudied, apart from anchor damage to coral reefs (e.g., Rogers and Garrison 2001). This leaves a significant gap in our knowledge. Cruise ships may be even more likely to have an impact on the local environment due to their large size, for example, by disturbing seafloor habitats in shallow waters during manoeuvring and anchoring (Broad et al. 2020). A lack of baseline data from before tourism in cruise ships became widespread may have allowed the impacts on marine life to go undetected so far.

1.2 Increasing traffic in Akaroa Harbour

Vessel traffic has increased substantially in Akaroa Harbour, an inlet on the south-facing portion of Te Pātaka-o-Rākaihautū/Banks Peninsula. Stone and Yoshinaga (2000) reported a doubling in recreational vessel traffic during the 1990s. Summertime vessel traffic has doubled again at least since then (Carome 2021).

The largest proportion of this increase is commercial dolphin tourism (Carome 2021). Beginning with a daily natural history tour in 1985 (Martinez 2010), dolphin-targeted tourism at Akaroa Harbour grew to six operators permitted to run up to 31 trips each day.

Eighteen trips provide swim-with-dolphin experiences and 13 provide dolphin watching (Tom MacTavish, pers. comm.). Between 2005 and 2008, commercial tour vessels accounted for 22% of all vessel traffic in Akaroa Harbour and 70% of interactions with dolphins (Martinez 2010).

Cruise ships also contributed to the increase in vessels in Akaroa Harbour. Following earthquake damage to the port in Whakaraupō/Lyttelton Harbour in Ōtautahi/Christchurch, annual cruise ship visits to the town of Akaroa increased more than fourfold during the summer of 2011-2012 (Fig. 1). Cruise ship visits remained high until they decreased during the COVID-19 pandemic.

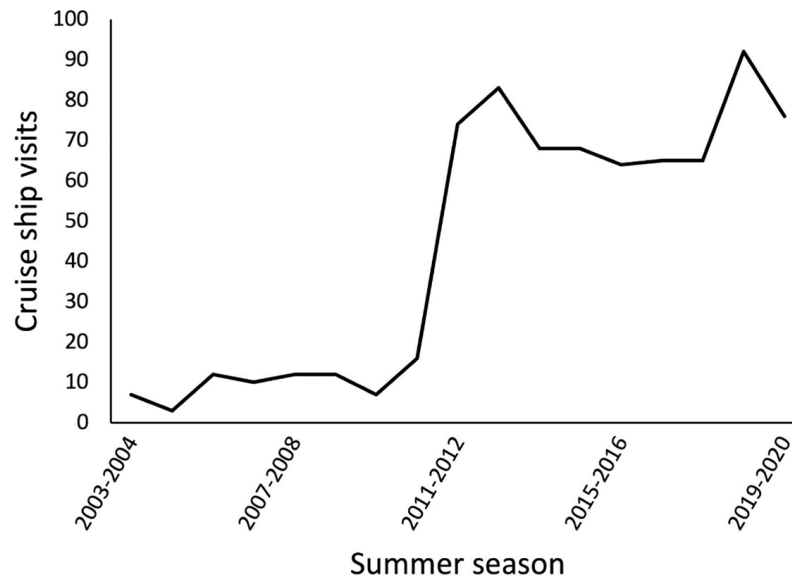


Figure 1. Number of cruise ship visits to Akaroa Harbour in each summer season (October to April) from October 2003 to April 2020 (from Wilson et al. 2015).

1.3 Vulnerability of Hector’s dolphin to vessel traffic

The growth in both cetacean tourism and the cruise ship industry is likely to affect the Hector’s dolphin (*Cephalorhynchus hectori*). Hector’s dolphin is found only in coastal waters around Aotearoa New Zealand. It is classified as ‘Endangered’ by the International Union for Conservation of Nature, IUCN (Reeves et al. 2013) and ‘Threatened – Nationally Vulnerable’ by the Department of Conservation, DOC (Baker et al. 2019).

Hector’s dolphins have high site fidelity, meaning they tend to return to previously visited locations (Rayment, Dawson, Slooten, Bräger et al. 2009). They are largely found in near-shore waters. This increases their exposure to anthropogenic activities. Some individuals are likely to be repeatedly exposed to human pressures (e.g., Martinez et al. 2011; Rayment, Dawson, Slooten, Bräger et al. 2009), potentially leading to cumulative impacts.

The effects of vessel traffic on Hector’s dolphins were previously examined in the Catlins and at Te Pātaka-o-Rākaihautū/Banks Peninsula. Bejder et al. (1999) used land-based theodolite tracking to measure the reaction of Hector’s dolphins to a single boat used for tourism at Porpoise Bay in the Catlins. Dolphins were not displaced by the vessel, but changed their reactions to it as encounters progressed and altered their group dispersion. In a similar study by Green (2003), the effects of a vessel on dolphin behaviour were minor and variable, including differences in dispersion and behavioural budgets¹. At Akaroa Harbour, Martinez (2010) examined the short-term behavioural responses of Hector’s dolphins to vessels from

¹ A behavioural budget is the time allocated to specific activities, such as travelling, diving, feeding and socialising.

land-based theodolite observations and data gathered opportunistically from tour vessels. Dolphins altered their behavioural budget, spending significantly less time diving and travelling when vessels were within 300 m. Dolphins also showed significant behavioural changes when swimmers were present in the water (Martinez et al. 2011).

Such changes have the potential to alter foraging success (Martinez 2010). A previous study at Akaroa Harbour by Nichols et al. (2001) found an inverse, although not statistically significant, relationship between the number of dolphins and vessels. The research to date suggests that the presence of vessels can alter the behaviour of Hector's dolphins and may have further impacts on their distribution. No research is currently available on whether the effects on behaviour have broader biological consequences, for example on survival or reproductive rates.

1.4 Previous estimates of abundance of the Hector's dolphin population

Modelling of the whole Hector's dolphin population in Aotearoa New Zealand shows up to 80% of the population was depleted since the early 1970s. This was mainly due to individuals becoming entangled in the nets used in gillnet and trawl fishing activities (Dawson 1991; Slooten 2007). A study showing a significant loss of genetic diversity in contemporary samples (1988–1998) compared to historic samples in museums (1870–1987) also revealed a serious decline in the east coast population of Te Waipounamu/the South Island (Pichler and Baker 2000).

The Hector's dolphin population at Te Pātaka-o-Rākaihautū/Banks Peninsula, has been studied since 1984 (Slooten and Dawson 1988). A boat-based line-transect survey in 1998 out to four nautical miles offshore yielded the first statistically robust abundance estimate of 897 individuals. This estimate is considered robust as the observations had a relatively low statistical variability, shown by a low coefficient of variation, CV, of 0.28 (Dawson et al. 2004). Analysis of photo-ID capture-recapture data gathered over a similar area during the 1990s produced an estimate of 1,119 dolphins (CV = 0.21) in 1996 (Gormley et al. 2005).

The next systematic population survey was a series of aerial line-transect surveys in 2013. These covered a wider offshore zone, and yielded an estimated 2,567 individuals for the area out to four nautical miles from the coast, and 5,025 out to 20 nautical miles (McKenzie and Clement 2014).

Over time, protection for the dolphins from gillnetting, and to a lesser extent from trawling, changed dramatically. Creating the Banks Peninsula Marine Mammal Sanctuary in 1988 provided the first protection from gillnetting (Dawson and Slooten 1993). Demographic modelling based on photo-ID data suggested that this protection improved survival rates, but probably not sufficiently to result in growth of the population (Gormley et al. 2012). The protected area expanded in 2008 and 2020, and now includes some restrictions on inshore trawling (see Slooten and Dawson 2021 for details). The effects of the additional protection have not been investigated yet.

1.5 Aims of this study

It is timely to reassess the status of the Hector's dolphin population at Te Pātaka-o-Rākaihautū/Banks Peninsula to:

- determine the impact of the recent increases in vessel traffic in Akaroa Harbour, including the effects of the increases in dolphin tourism since the 1980s and the jump in cruise ship visits after 2012
- support evaluations of the sustainability of tourism activities focused on the dolphins in context with their present conservation status (classified as 'Endangered' and 'Threatened - Nationally Vulnerable' by the IUCN and DOC respectively)
- inform DOC decisions about a current moratorium on issuing permits for dolphin-focused tourism around Akaroa Harbour (named 'the Akaroa Operational Area' internally to DOC) which expires in 2026.

In this report, the impacts of vessel traffic on Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula will be determined by addressing the following questions:

- Is the population of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula increasing, decreasing or stable?
- What proportion of this dolphin population interacts with tourism in Akaroa Harbour?
- Have dolphins changed their use of the Akaroa Harbour due to tourism?

The findings for this last research area were presented in previously published work and are included in this report for completeness. See Carome (2021) and Carome et al. (2022, 2023a, 2023b) for details.

2. Methods

To determine the impact of vessel traffic on Hector's dolphins at Te Pātaka-o-Rākaihautū /Banks Peninsula, we used:

- survey data (photographs and observations) collected between 1984 and 2022 by researchers at the University of Otago
- passive acoustic monitoring in Akaroa Harbour in 2019 and 2020
- time-lapse photography from an automatic camera system at Nine Fathom Point in Akaroa Harbour in 2019 and 2020
- GPS tracking data from vessels providing dolphin-watching experiences
- the locations of medium and large cruise ships moored in Akaroa Harbour
- statistical modelling of the datasets derived from these observations.

In this section, we describe the data collection and provide a high-level overview of the established statistical methods we used to model the population of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula.

2.1 Surveys by boat between 1984 and 2022

Surveys were carried out by boat at Te Pātaka-o-Rākaihautū/Banks Peninsula each summer season between 1984 and 2022, except during the summer seasons of 1998–1999 and 1999–2000.

These surveys:

- looked for and approached groups of dolphins – a group is a number of dolphins within approximately five body lengths of each other, closely associating and engaged in similar activities (Gowans et al. 2001; Constantine et al. 2004)
- took photographs of dolphins with markings suitable for identification of individuals
- recorded each encounter for later modelling to determine the distribution of the population of Hector's dolphins in the area.

The surveys were conducted from small (< 7 m long) outboard-powered boats, following a standard protocol (Slooten et al. 1992; Bräger 1998; Rayment, Dawson, Slooten, Bräger et al. 2009). Courses were steered while referring to known landmarks and/or GPS. Surveys were carried out at speeds of 10–15 knots.

All surveys were carried out when conditions at sea were three or less on the Beaufort scale (i.e., wind speeds \leq 10 knots), with swell heights less than 2 m, and during good visibility. Decisions on where to survey were largely based on weather and sea state, but also to boost coverage in areas least surveyed recently.

Details of each survey were recorded on either:

- a Hewlett Packard 200LX palmtop computer running custom written software and connected to the GPS, or
- a tablet running a custom program in the *Cybertracker* app.

The position of the vessel was logged at least every two minutes creating a tracking file which was downloaded at the end of each day.

2.1.1 Location of the surveys

Most of the surveys were conducted within 1 km of the coastline between Sumner Head and Birdlings Flat (Fig. 2). The areas to the north of Te Pātaka-o-Rākaihautū/ Banks Peninsula, southwards to the Rakaia River, and offshore to 10 km from the coast were surveyed less frequently.

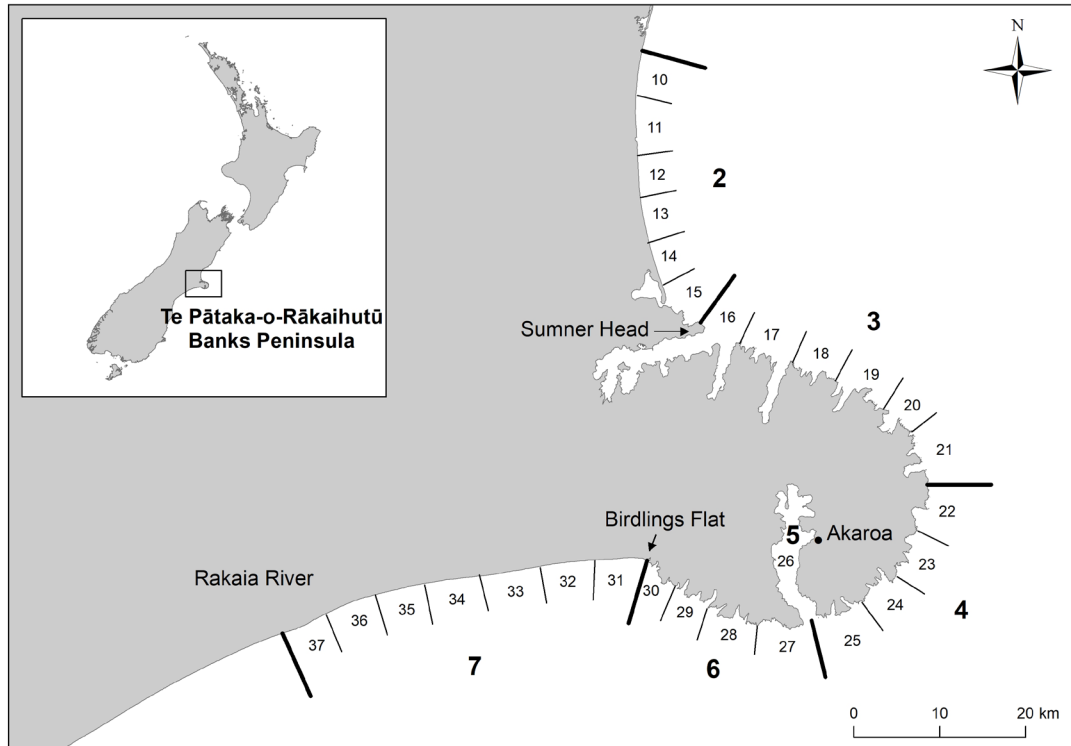


Figure 2. Map of the study area at Te Pātaka-o-Rākaihautū/Banks Peninsula showing numbered sections (10–37) and broader areas (2–7), which were used to summarise survey effort. This stretch of coastline contains the Banks Peninsula Marine Mammal Sanctuary.

Surveys outside of Akaroa Harbour and around the peninsula were typically alongshore and approximately 400 m from the coast. This distance was chosen to maximise the encounter rate with Hector’s dolphins (Dawson and Slooten 1988). These surveys were usually conducted in one direction along the coast. Occasionally, if weather and sea conditions were favourable, return surveys were completed between 1 and 5 nautical miles offshore.

Within Akaroa Harbour, the survey route followed a set zigzag course (Dawson 1991; Fig. 3).

The distribution of effort in the survey area was summarised by dividing the coastline into sections approximately 6 km in length (Fig. 2). The amount of surveying work undertaken was tracked by counting the number of times that that the survey was carried out in each section. Effort was further summarised by grouping sections into six broader areas (Fig. 2).

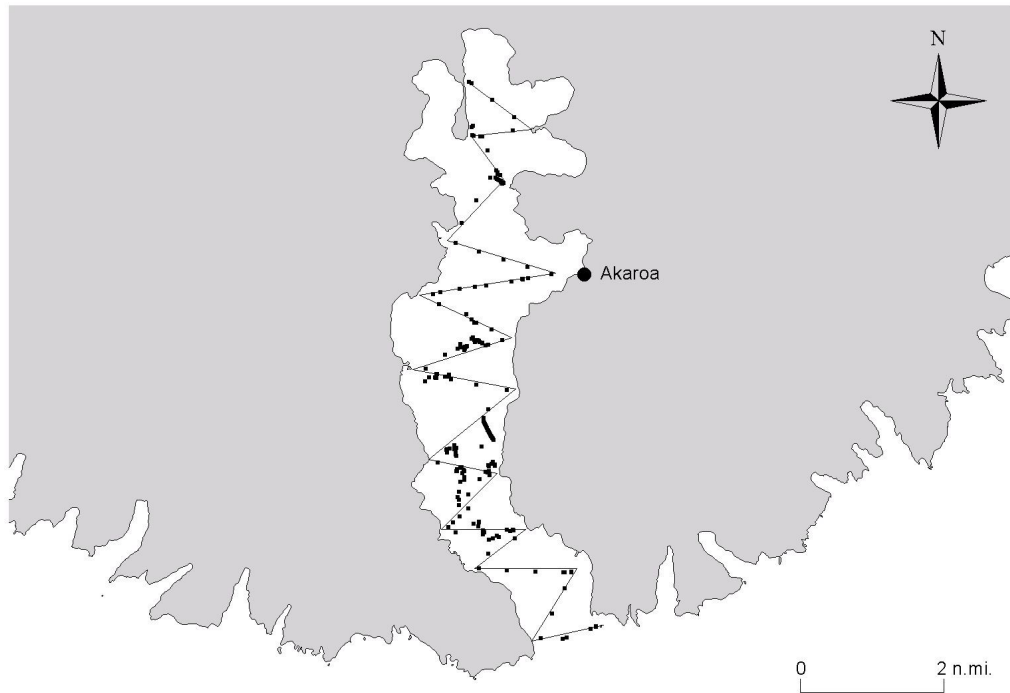


Figure 3. A typical survey of Akaroa Harbour carried out along the set course. The dots are GPS fixes every two minutes. The solid line is the zigzag route. The GPS fixes show the deviations from the route when dolphins were encountered.

2.1.2 Carrying out the surveys

During surveys, at least two observers were constantly looking for dolphins. The search effort was divided so that at least one observer concentrated on the track-line (the route being taken by the boat) and the port (left) side of the survey vessel. At least one other observer concentrated on the track-line and the starboard (right) side.

When a group of Hector's dolphins was sighted, the course of the survey vessel was adjusted to approach the group and the vessel was slowed to idle speed.

Details of each encounter were recorded on the palmtop computer or tablet. The start time and GPS position of each encounter were recorded via a single keystroke when the dolphin group was first approached. The end time and position were recorded when the encounter ended. This generated an 'encounter' file which was downloaded at the end of each day.

Other details recorded for each encounter included: the group size, the number of calves present, the water depth (noted from onboard echosounders), sighting conditions, group behaviour, notable individual behaviours, presence of any identifiable individuals and any other relevant information.

Calves (up to one year old) were identified by their small body size, presence of foetal fold marks, and consistent close association with a large adult, presumed to be their mother (Smolker et al. 1992; Slooten and Dawson 1994; Grellier et al. 2003).

Each member in the dolphin group was examined to assess whether it had any markings suitable for identification in photographs. If a dolphin had identifiable markings, it was photographed. Ideally, photographs were taken of both sides of the dolphin at close range (< 10 m).

Once all suitable markings in a group were photographed, the vessel left the group at low speed and the original course was resumed as close as possible to the point where the boat left the planned survey route.

During 1992 to 1996 and in 2016, ‘random photography’ was used alongside the routine photographic surveys (see Wickman et al. 2021a for a detailed description). This involves photographing any individual that surfaces within range of the photographer (not just marked individuals). This allows an estimate of mark rate, the proportion of dolphins in the population with distinctive markings.

When a dolphin group did not actively approach the boat, the group was approached a maximum of three times before the encounter was abandoned. This approach aimed to minimise disturbance to dolphins that were avoiding interaction with the boat. When a dolphin group was seen but could not be located again, the boat left after a 5-10 minute search.

The way these surveys are carried out has remained virtually the same since the beginning of the project in 1984. Minor changes were largely due to using new technologies as they became available and affordable. For example, the locations of the surveys and sightings have been logged by GPS since 1990, and slide-film photography was replaced by digital photography in 2001. Also starting in 2001, two boats were used regularly during summer, so that two areas could be surveyed simultaneously when weather conditions allowed.

2.1.3 The catalogue of Hector’s dolphin photographs

Protocols for setting up and maintaining the University of Otago catalogue of photographs of Hector’s dolphins are well established. See Slooten et al. (1992), Bräger (1998) and Rayment (2008) for more information.

The catalogue is divided into sections containing individuals with similar markings to facilitate filing and matching images. Before 2001, the catalogue consisted of printed slides stored in loose-leaf binders. The catalogue has since been digitised and new digital images are added as .jpg files.

Each dolphin is given a unique alphanumeric code in the catalogue according to the mark type and similarity to other individuals in that section. Each individual is also assigned a category between one and three according to how obvious its mark is and how likely it is to be noted in the field.

Slooten et al. (1992) defined the three categories.

- Category 1 individuals have such obvious marks that they are unlikely to be missed in the field, and have an excellent chance of being identified from photographs of each encounter.
- Category 2 individuals also have obvious identifying marks, and are unlikely to be misidentified from good photographs. However, they are more likely to go unnoticed in the field or to be difficult to identify from photographs taken at less than ideal angles.
- Category 3 individuals have more subtle marks which are useful for distinguishing animals while observing behaviour or group interactions, but may be easily missed in the field and/or are difficult to positively identify from photographs.

The catalogue, and storage of associated sighting information, has evolved over time. In its current form, the data are housed in a relational database in Microsoft Access. This includes data on individual dolphins, encounters and surveys.

2.1.4 Matching photographs to individuals

Each new image was examined to check if it met the following quality criteria;

- Images were rejected if they were not in sharp focus and correctly exposed.
- The dolphin needs to be close enough in the image and in the correct orientation to show the mark of interest (usually perpendicular to the view of the photographer).

Meeting these quality criteria ensures that all obviously marked individuals have approximately the same probability of being identified. This means there is no bias in the probability that an individual will be captured in a photograph, the ‘capture probability’ (Stevick et al. 2001; Urian et al. 2015). This is especially important for mark-recapture analyses (identifying individuals that were previously photographed).

Images of sufficient quality were then compared to the existing information in the catalogue.

- If a dolphin already existed in the catalogue, its identity was noted, and the details of its encounter were recorded.
- If the new image was superior to the existing catalogue image, or added new information (e.g., a previously unphotographed side or an additional mark), the image was added to the catalogue.
- If the individual was not already in the catalogue, it was assigned an alphanumeric name according to its mark type and added to the catalogue.

All matching was done by eye, in consultation with at least one other experienced researcher. All resightings of individuals were confirmed by comparing images taken during surveys with those stored in the catalogue.

2.2 Automatic monitoring during 2019 and 2020

To investigate the effects of vessel traffic within Akaroa Harbour, we gathered passive acoustic monitoring data to measure the distribution of Hector’s dolphins in the vicinity over the summer of 2019–2020. We compared these with data on vessel traffic. The vessel data were obtained in two ways, through:

- an automated camera system overlooking Akaroa Harbour, and
- satellite tracking of tour vessels and cruise ships.

We summarise the methods here; full details are available in Carome et al. (2023a).

2.2.1 Passive acoustic monitoring

The aim of passive acoustic monitoring was to detect potential changes in the distribution of dolphins compared with vessel traffic parameters.

An automatic echolocation detector (T-POD v.5 number 560, Chelonia Ltd.) was moored in Akaroa Harbour off Nine Fathom Point from 9 November 2019 to 3 May 2020, within view of an automated camera system (Fig. 4). The T-POD was suspended five meters below a moored surface buoy with a dive weight affixed to ensure vertical orientation. The waters off Nine Fathom Point were chosen due to the well-documented overlap between Hector’s dolphins and vessel traffic at this site (Nichols et al. 2001; Martinez 2010; Brough et al. 2019). The T-POD was serviced every three to four weeks (data downloaded, batteries replaced, and fouling removed). A single failed deployment meant data were not collected between 10 January 2020 and 6 February 2020.

In each deployment of the T-POD, the first five scans were optimised to detect Hector’s dolphins (i.e., specifying a target filter frequency of 130 kHz). The sixth scan was optimised to detect other species of dolphin. Identical settings were used in previous studies on Hector’s dolphin habitat use (e.g., Rayment et al. 2011; Dawson et al. 2013; Leunissen et al. 2019). At similar settings, T-PODs have had an effective detection radius of 198 m for Hector’s dolphins, with the most distant known detection being 431 m from the T-POD (Rayment, Dawson, Slooten 2009).

We extracted and analysed data on dolphin echolocation clicks using the software TPOD.exe (v.8.24, Chelonia Ltd). The software's train detection algorithm classifies sequences of clicks (trains) based on the probability of being produced by cetaceans. The categories CET HI and CET LO reliably consist of Hector's dolphin clicks, and conservatively estimate their presence (Rayment, Dawson, Slooten 2009). These categories are collectively termed 'CET ALL' and were used in this study.

Click data were extracted as detection positive minutes (DPM) between 08:00 and 18:00 for each day. DPM is the standard metric for studying habitat use with both T-PODs (e.g., Leunissen et al. 2019) and C-PODs (a similar passive acoustic detector, e.g., Roberts and Read 2015). DPMs represent the number of minutes per day in which at least one dolphin click train was detected on scans 1-5 and categorised as CET ALL. We chose an observation period of 08:00 to 18:00 as it includes the hours when the full suite of vessel-traffic variables could be recorded reliably. In addition, temporal autocorrelation of the DPM response variable was reduced to acceptable levels by using this time period.

2.2.2 Visual monitoring with time-lapse photography

We mounted an automatic camera system at a cliff-top station at Nine Fathom Point at 123 m above sea level (Fig. 4). During 1-9 December 2019, a Canon DSLR (Canon EOS Digital Rebel XS; 10.1MP, Canon EF-S 18-55 mm f/3.5-5.6) and Harbortronics housing and controller system (Digisnap 2000, Harbortronics Inc.) were installed and programmed to capture a still image every two minutes continuously. On 9 December, we replaced the camera and controller with a Micro Four Thirds camera (Olympus E-PM2; 16MP, DJI 15mm f/1.7 Prime Lens) and a custom-built time-lapse controller to improve image resolution and reliability.

We mounted the system on a steel frame built for previous research (Rayment et al. 2018) and powered by an 11.1 V 9AH battery charged by two solar panels affixed to the top of the frame. We programmed the controller to power the camera on, capture a still image, and power the camera off, every two minutes from 06:00 to 20:30 each day. This system was serviced every three weeks and ran faultlessly until it was dismantled on 14 May 2020.

We conducted a set of calibration exercises to determine the extent of the study site. We mounted a second Olympus E-PM2 camera with an identical 15 mm lens on a tripod approximately 0.5 meters directly seaward of the automated camera. An onshore observer directed a 6.6 m research vessel (RV) along the edges of the field of view via VHF radio, to determine their boundaries. Seventeen GPS fixes were logged from the RV by pressing a hotkey on the onboard palmtop computer (Hewlett Packard 200LX) connected via serial port to a GPS (Garmin GPS 196).

To determine the seaward extent of the study site in images, the RV was manoeuvred 1 km away from the surface buoy of the T-POD, determined by GPS, in line with the surface buoy and automated camera (Fig. 4). At this location, a GPS fix was logged, and an image was captured on the tripod-mounted camera. An overlay derived from this exercise was applied to each image for analysis.

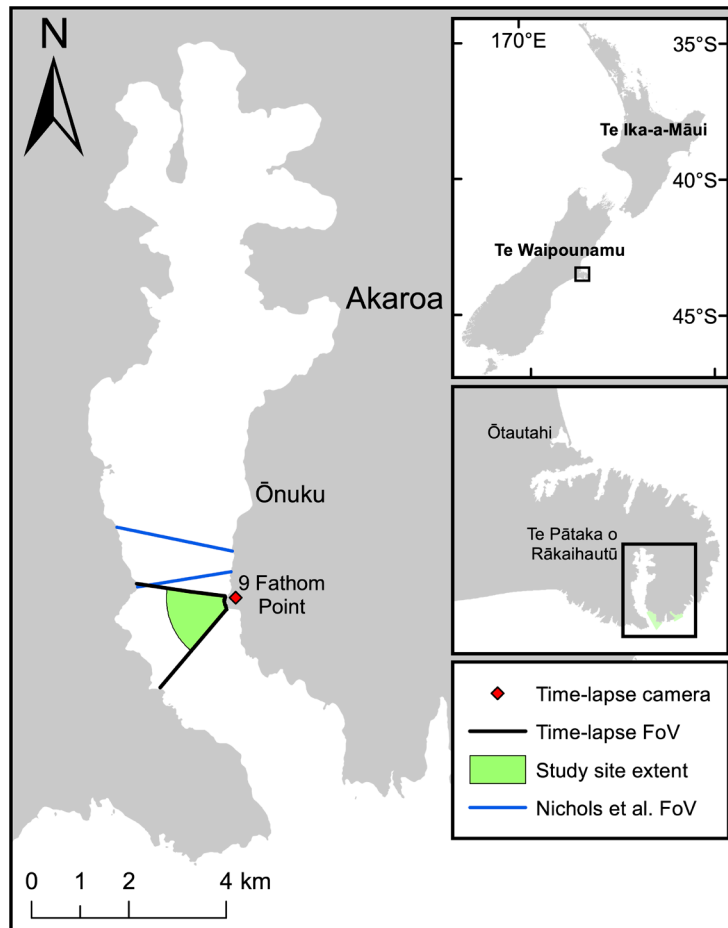


Figure 4. Map of Akaroa Harbour showing: the study site used by Nichols et al. (2001) to monitor vessel traffic in 1999 and 2000, the location (43°51.210' S, 172°56.469' E) and field of view (FoV) of the automatic time-lapse camera station, and the area of the harbour visible to the camera (shown in green), which was used to monitor vessel traffic from December 2019 to May 2020. Reproduced from Carome et al. (2023a).

To maintain consistency with seasonal changes in daylight hours, we only included images taken between 08:00 and 18:00 in our analyses. Each image was scored based on visibility (0 to 5), with zero representing a completely blurred or obscured image, and five representing a clear, crisp and unobstructed image. We only included images with visibility scores of three or more in our analyses. One person then recorded the number of vessels present in each image, categorised as motor or nonmotor (i.e., kayaks or boats under sail).

The maximum number of images taken in a day (08:00 to 18:00) was 300 (120 s intervals giving 30 images per hour over 10 hours). We excluded days from the analysis if they had fewer than 270 acceptable images or contained one or more hours with fewer than 10 acceptable images. We calculated the number of unique vessels during each hour of observation.

For days with 270 or more acceptable images, a pro-rata correction was applied to any given hour if any images were not of an acceptable quality:

$$V_{corrected} = \frac{V_o}{I_a} \times I_{max},$$

where $V_{corrected}$ is the pro-rated number of vessels in a given hour, V_o is the observed number of vessels from all acceptable images in that hour, I_a is the total number of acceptable images in that hour, and I_{max} is the maximum possible number of images in an hour (in this case 30).

2.2.3 Tracking of tour vessels and cruise ships

We installed custom-built BeechTrack GPS tracking devices (TrackMe NZ) on all vessels permitted to view dolphins at Akaroa Harbour. These devices continuously logged the vessel location at 10 s intervals. The GPS tracks showed the number of dolphin tourism trips per day of one hour or longer.

We determined the number of cruise ships anchored at Akaroa each day from automatic identification system (AIS) tracks for each ship. We categorised the cruise ships based on passenger capacity as: small (< 500 passengers), medium (between 500 and 2,000 passengers) or large (> 2,000 passengers). Small cruise ships carry relatively few passengers (on average = 146 ± 7 standard error, SE) compared to medium and large ships (mean = $2,398 \pm 119$ SE).

The small cruise ships anchored further away from our study site. Medium and large cruise ships generally anchored close to our study area. We assumed that these larger cruise ships were likely to present a greater risk of impact on Hector's dolphins at Nine Fathom Point. This assumption was primarily based on the proximity of their anchorages, the size of vessels, their likelihood to produce greater source levels of noise at anchorage, and the influence of passenger capacity on the number and frequency of movements of tender vessels (other vessels servicing the cruise ships). Thus, we only examined medium and large cruise ships in our models.

2.3 Statistical analysis

We followed the Bayesian modelling approach used by Gormley et al. (2012) on an earlier version of the same dataset to estimate changes in the apparent survival rate of Hector's dolphins over time. This uses a modified version of the Cormack-Jolly-Seber (CJS) model (Cormack 1964; Jolly 1965; Seber 1965). CJS models are used widely to analyse animal capture histories (in this case the 'capture-recapture' data of the dolphins with identifying marks). As we had included additional data up to 2020 that was not in the Gormley et al. (2012) study, we defined a third period for estimating the mean survival rate (2008–2020) and included other modifications relating to modelling the mean probability of survival. For more details, see Appendix 1.

Temporal and spatial variations in the reproductive output of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula were investigated by fitting linear models to the proportions of calves (up to one year old) sighted in the population (e.g., Wells and Scott 1990). This is covered in detail in Appendix 2.

The photographic capture-recapture data were also used to estimate changes in population size over time (see Appendix 3 for details). The total population size was derived by estimating the number of marked dolphins each year using a CJS model, and scaling up according to an estimate of mark rate obtained using the 'random' photography. We analysed a subset of the data to make sure we minimised the biases that might occur due to surveying in different places over time (minimising the 'capture heterogeneity'). We also allowed different estimates of mean survival rate for the periods corresponding to different levels of spatial protection. We chose this approach because fisheries bycatch is likely to be the greatest impact on Hector's dolphins (Dawson 1991, Pichler and Baker 2000, Slooten 2007) and therefore the main driver of changes in survival rate.

We estimated the proportion of the Hector's dolphin population using Akaroa Harbour by dividing the number of marked individuals photographed in the harbour by the total number of individuals in the University of Otago catalogue of photographs of dolphins in Te Pātaka-o-Rākaihautū/Banks Peninsula. Appendix 4 provides details about this analysis.

The potential effect of cruise ships on Hector's dolphins in Akaroa Harbour, and any variation in their use of the harbour, was visualised with kernel density estimation (KDE; Worton 1989). This compared KDEs of the distributions of dolphins during 2000–2011 and 2012–2020, the periods before and after the four-fold increase in number of cruise ships anchoring in the middle of the harbour (Appendix 5).

Finally, we used generalised additive models (GAMs; Hastie and Tibshirani 1990) of acoustic detection rates at Nine Fathom Point to determine the effect of vessel traffic on fine-scale distribution of Hector's dolphins (Appendix 6). This used vessel traffic parameters estimated using the automated camera system, GPS tracks of tour vessels, and AIS tracks of cruise ships, as well as environmental and temporal parameters thought to influence distribution of dolphins.

3. Results

3.1 Estimating survival rate

A total of 582 Hector’s dolphins with category 1 and 2 marks (obvious identifying marks) were included in the analysis for estimating the apparent survival rate. These were analysed over three time periods: 1986–1989, 1990–2008 and 2009–2019, to reflect the differences in management of the area after the Banks Peninsula Marine Mammal Sanctuary was established and the later restrictions on gillnetting and trawling.

The annual estimates of survival rate showed significant year-to-year variation (Fig. 5). Although the mean estimates of survival rate increased between successive management periods, there was significant overlap between their 95% HDIs² (Fig. 6).

The mean apparent survival is given by the median of the posterior distribution of $\text{expit}(\mu_\phi) = \phi$ for each period (see Appendix 1 for details). This was estimated as:

- 0.901 (95% HDI 0.817–0.961) for 1986–1989
- 0.907 (95% HDI 0.879–0.941) for 1990–2008, and
- 0.923 (95% HDI 0.881–0.964) for 2009–2019 (Table 1).

Mean survival in the latest management period (2009–2019) was higher than mean survival before protection from the Banks Peninsula Marine Mammal Sanctuary (1986–1989) in 72.3% of samples. The modelling suggests that the mean survival rate of Hector’s dolphins increased 2.2% between 1986–1989 and 2009–2019.

Table 1. Summary of key parameters estimated by the survival-rate model. Differences in mean survival between different periods ($\Delta\mu_\phi$) are on the logit scale: $\phi_{2,1}$ for comparing (1990–2008) with (1986–1989), $\phi_{3,2}$ for comparing (2009–2019) with (1990–2008), and $\phi_{3,1}$ for comparing (2009–2019) with (1986–1989). The probability of an increase in the dolphin survival rate was taken as the percentage of samples where the difference is greater than 0.

PARAMETER	MEDIAN (50%)	2.5%	97.5%	MEAN	SD	PROBABILITY OF INCREASE ($\Delta\mu_\phi$)
$\phi_{1986-1990}$	0.901	0.817	0.961	0.898	0.037	
$\phi_{1990-2008}$	0.907	0.879	0.941	0.908	0.0157	
$\phi_{2009-2019}$	0.923	0.881	0.964	0.923	0.0211	
σ_ϕ	0.440	0.0405	1.05	0.463	0.263	
$\Delta\mu_{\phi_{2,1}}$ (logit scale)	0.0732	-0.964	0.981	0.0559	0.489	0.566
$\Delta\mu_{\phi_{3,2}}$ (logit scale)	0.204	-0.436	1.02	0.230	0.368	0.735
$\Delta\mu_{\phi_{3,1}}$ (logit scale)	0.281	-0.731	1.34	0.286	0.521	0.723

0

² A statistical measure of the confidence in the modelling. Estimates within the 95% highest density level (95% HDI) have a higher probability of being credible than points outside of this.

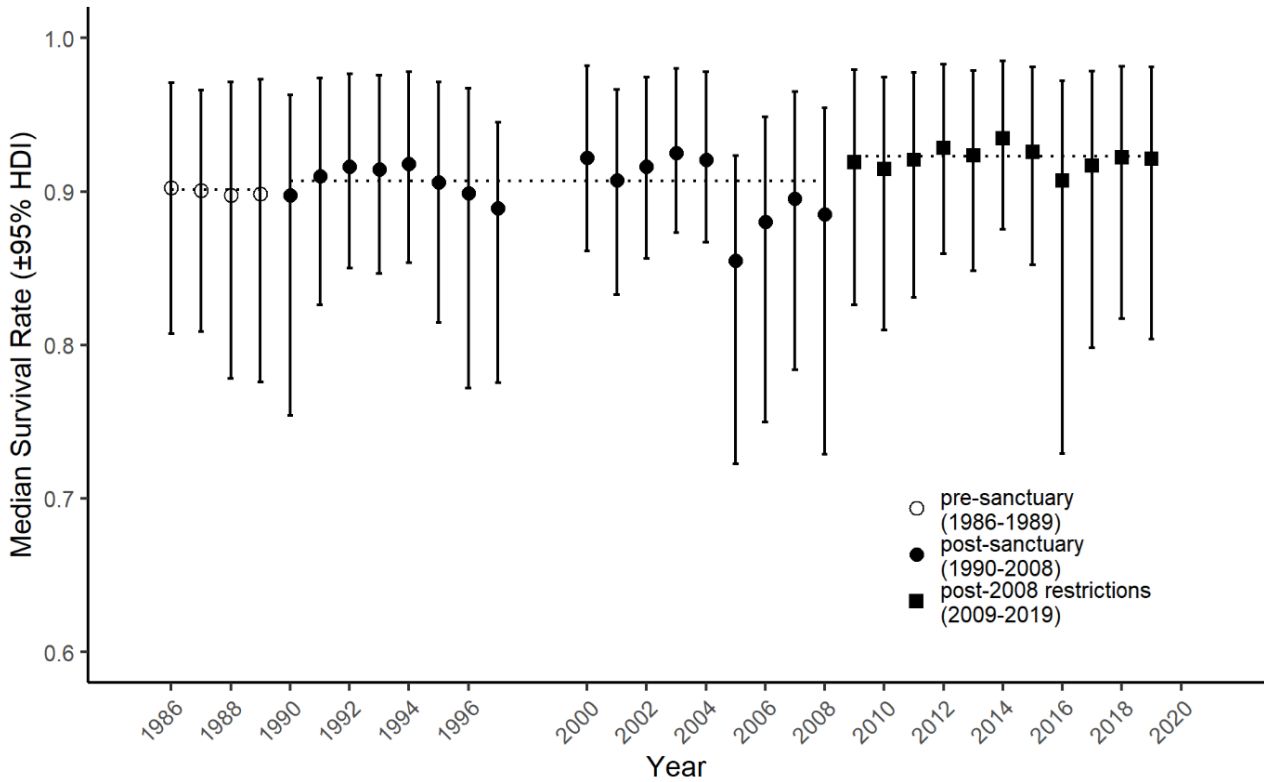


Figure 5. Annual median survival, with error bars representing the 95% HDI of the posterior. Dashed lines represent the mean survival estimate (see median column in Table 1) for each of the three management periods (1986–1989, 1990–2008 and 2009–2019).

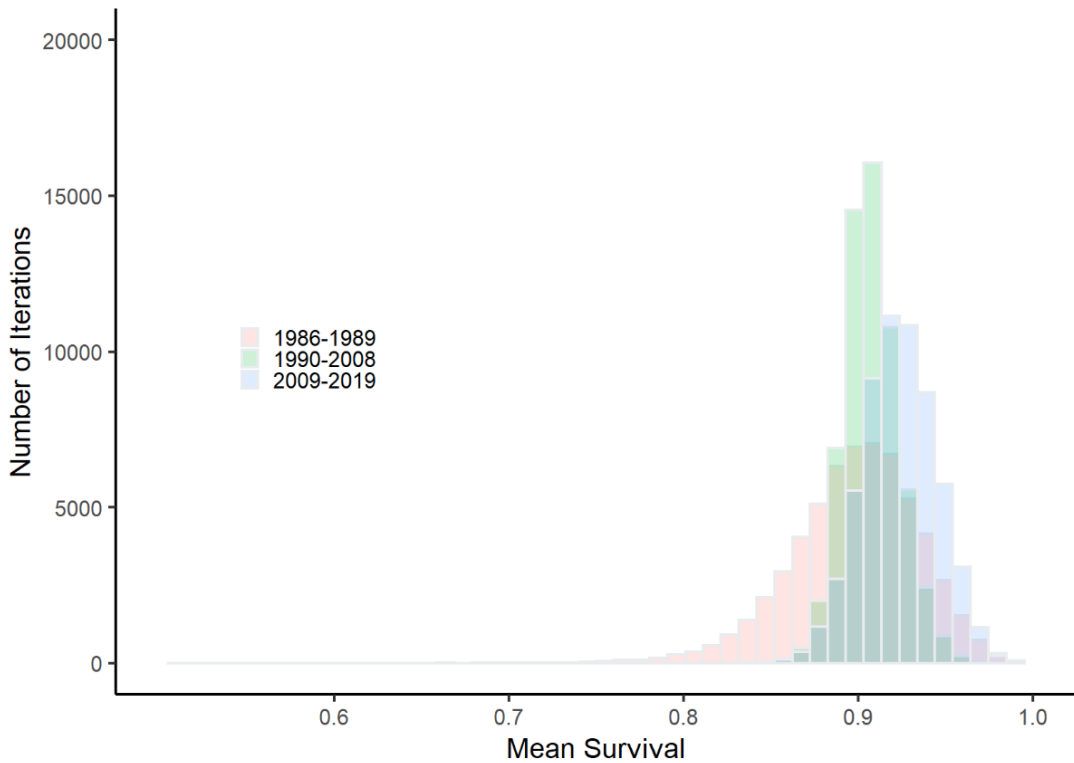


Figure 6. The overlaid distributions of mean survival for each of the three management periods.

3.2 Reproductive output of Hector’s dolphins

The reproductive output was investigated by fitting linear models to the proportions of calves (up to one year old) sighted in the population (e.g., Wells and Scott 1990).

For dolphins sighted between Summer Head and Birdlings Flat (Fig. 2), the annual proportion of calves in the population varied from 0.007 in 2020 to 0.084 in 1995 (mean = 0.042, SE = 0.003; Fig. 7). There was no evidence of a linear trend in proportion of calves over time ($F_{1,31} = 0.202$, $p = 0.656$). See Appendix 2 for details.

The effect of where the data were collected was statistically significant in the linear model. Area 3, on the north side of Te Pātaka-o-Rākaihautū/Banks Peninsula (Fig. 2), had the lowest proportion of calves, while area 6 on the south-west side had the highest (Table 2; Fig. 8). There was no evidence that the proportion of calves differed between Akaroa Harbour and the remainder of the survey area (paired t-test; $t = 0.810$, $p = 0.424$).

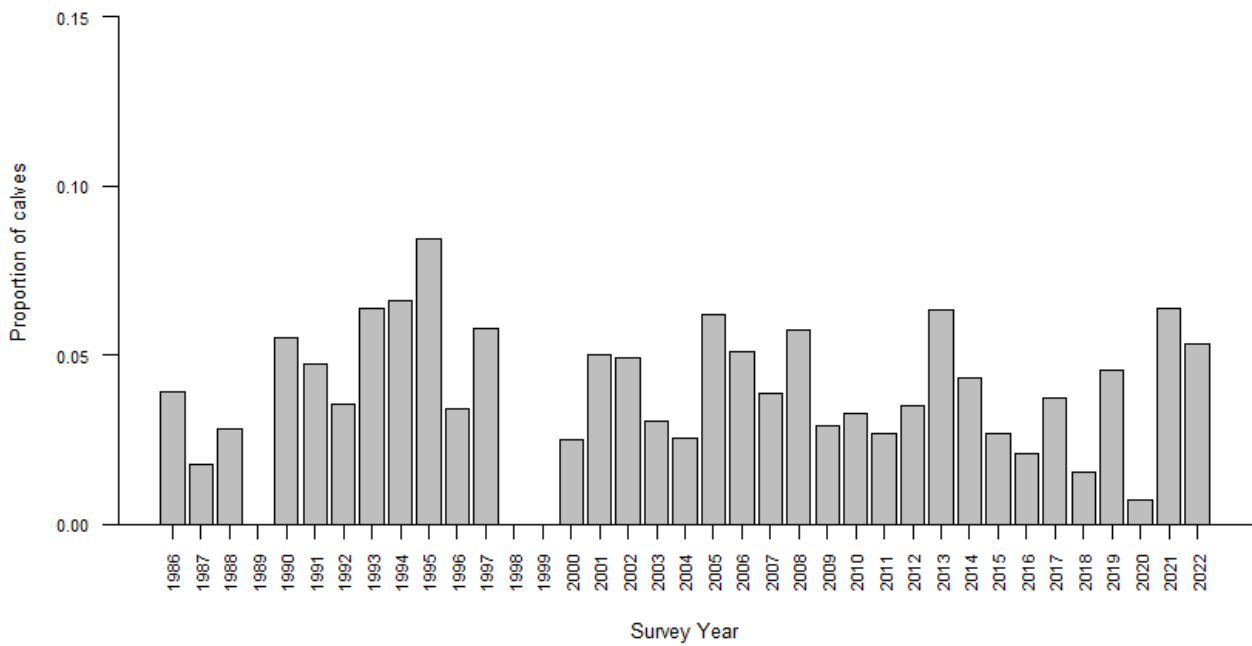


Figure 7. Annual variation in proportion of calves (up to one year old) encountered in the population of Hector’s dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula (1986–2022), using sightings from November to March each year. No data were available for 1989, 1998 and 1999.

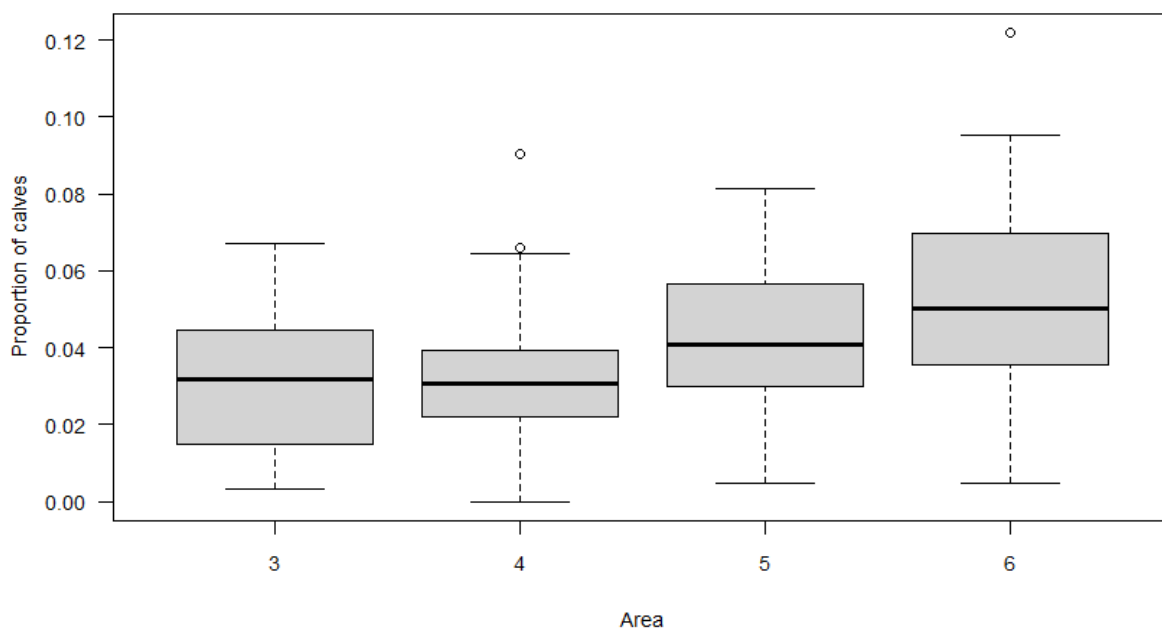


Figure 8. Boxplot showing proportion of calves (up to one year old) encountered in the Hector's dolphin population in four survey areas at Te Pātaka-o-Rākaihautū/Banks Peninsula (1986–2022). Fig. 2 shows the location of areas 3 to 6. Numbers of surveys: area 3 (north coast) = 2,686; area 4 (east coast) = 2,067; area 5 (Akaroa Harbour) = 5,424; area 6 (south coast) = 3,602.

Table 2. Results of the linear model comparing the annual proportion of calves (up to one year old) encountered in the Hector's dolphin population at Te Pātaka-o-Rākaihautū/Banks Peninsula in four different survey areas between 1986 and 2022. 'Year' is a continuous variable. 'Area' is a categorical variable referenced to Area 3.

VARIABLE	ESTIMATE	STANDARD ERROR	T-VALUE	P-VALUE
Year	-0.0003	0.0002	-1.265	0.208
Area – 4	0.0007	0.0058	0.113	0.910
Area – 5	0.0097	0.0056	1.747	0.083
Area – 6	0.0192	0.0056	3.409	0.001

3.3 Estimated population size from 2002 to 2020

A total of 351 Hector's dolphins with category 1 or 2 marks were included in the analysis for estimating abundance. Abundance of marked individuals between Birdlings Flat and Steep Head ranged from 98 in 2002 (95% HDI 72-149) to 126 in 2020 (95% HDI 71-191) (Fig. 9).

The results of a linear model fitted to these data suggested there was insufficient evidence for any trend over time in the number of marked individuals between 2002 and 2020 ($F_{1,17} = 0.038$, $p = 0.874$).

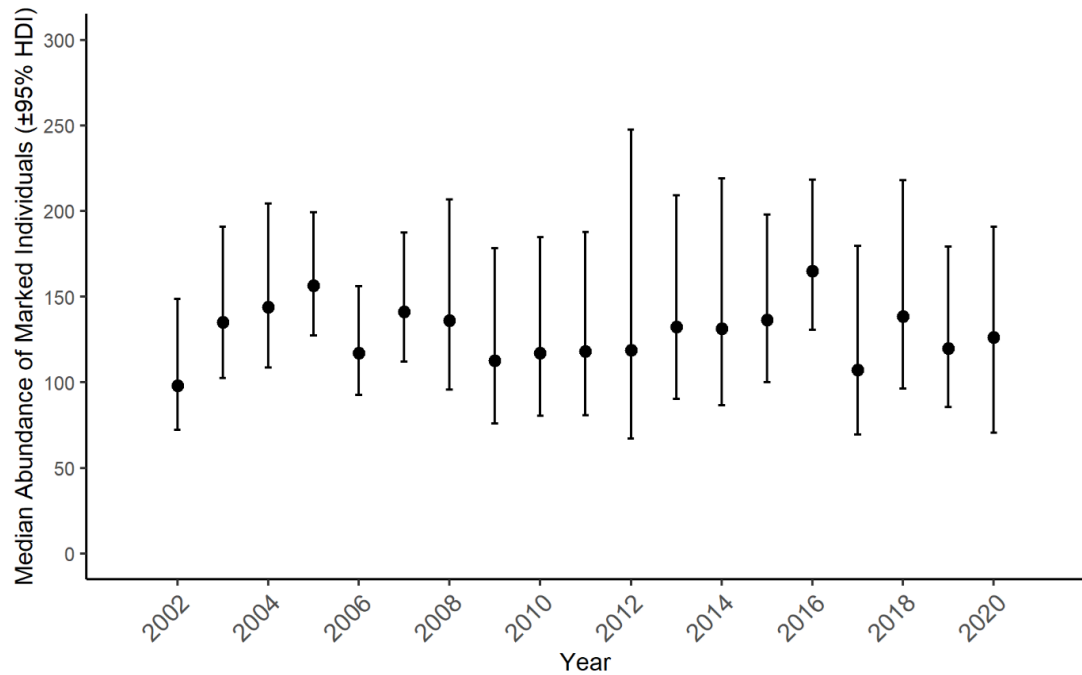


Figure 9. Annual median abundance for category 1 or 2 marked Hector's dolphins between Birdlings Flat and Steep Head from 2002 to 2020. The bars show 95% HDI.

Using 1,133 images from the random photography of 163 groups of dolphins, the mark rate (the proportion of dolphins in the population with distinctive markings) for dolphins with category 1 and 2 marks was estimated as 0.067 (95% HDI 0.049 - 0.091; Table 3). The estimated mark rate was then used to scale up the estimate of abundance from the marked proportion of the population to give yearly estimates of total population size for 2016 to 2020 (Fig. 10; Table 3). This subset of the abundance data was used because our most recent estimate of mark rate was from 2016.

The total abundance of Hector's dolphins, using the area between Birdlings Flat and Steep Head for the most recent study year, 2020, was 1,868 (95% HDI 988 - 3,154).

Table 3. Median, mean, standard deviation (SD) and the upper (97.5%) and lower (2.5%) HDI of samples from the posterior distributions for the mark rate ($\bar{\theta}_{2016}$) and the annual total abundance ($Ntotal_t$).

PARAMETER	MEDIAN	2.5%	97.5%	MEAN	SD
$\bar{\theta}_{2016}$	0.0674	0.0494	0.0908	0.0681	0.0106
$Ntotal_{2016}$	2,459	1,669	3,707	2,520	525
$Ntotal_{2017}$	1,599	941	2,896	1,681	505
$Ntotal_{2018}$	2,069	1,275	3,555	2,157	587
$Ntotal_{2019}$	1,784	1,134	2,944	1,851	468
$Ntotal_{2020}$	1,868	988	3,154	1,923	543

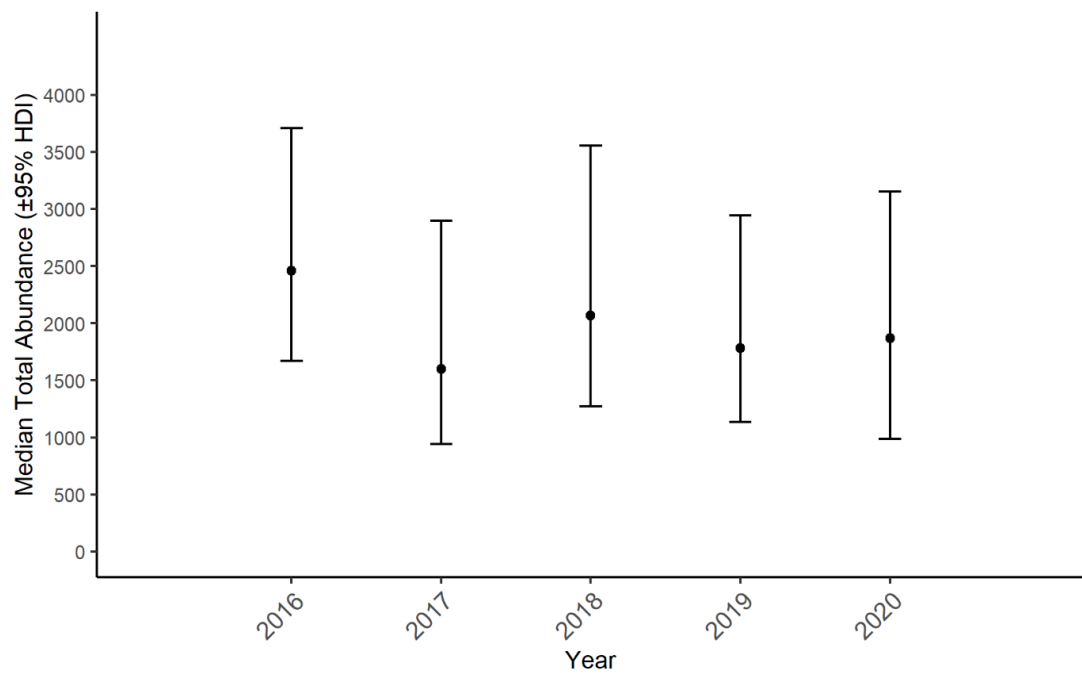


Figure 10. Annual median total abundance (adjusted for mark rate) for the period 2016–2020, between Steep Head and Birdlings Flat. The bars show 95% HDI.

3.4 The proportion of dolphins using Akaroa Harbour

Since 1985, 583 Hector’s dolphins with category 1 and 2 marks were sighted between Sumner Head and Birdlings Flat. Of these, 318 were seen at least once in Akaroa Harbour, a proportion of 54.5%. For the dolphins encountered since 2000, 224 were seen in Akaroa Harbour out of a total of 478, a proportion of 46.9%.

3.5 The effect of cruise ships on dolphin distribution in Akaroa Harbour

Between 2000 and 2020, there were 369 surveys that included set zigzag transects of Akaroa Harbour (Fig. 3). A total of 2,335 encounters with Hector’s dolphin groups were made across 8,732 km of surveys. Survey effort was distributed unevenly. The north-south midline of the harbour generally received the highest survey effort.

Dolphin distribution varied significantly between the two periods examined. In the period before expansion of cruise ship tourism (2000–2011), the core habitat for the dolphins was predominantly located in the middle of Akaroa Harbour (71%), encompassing or adjacent to the two primary anchorages for large cruise ships (Fig. 11). During the period after expansion of cruise ship tourism (2012–2020), the core habitat was concentrated in the eastern margin of the harbour, predominantly towards the seawards areas of the harbour (76%; Fig. 11). Areas in the middle of Akaroa Harbour near the designated anchorage locations for large cruise ships were no longer within the core habitat for Hector’s dolphins. This was based on 50% kernel density estimation (KDE). See Appendix 5 and Carome et al. (2022) for details.

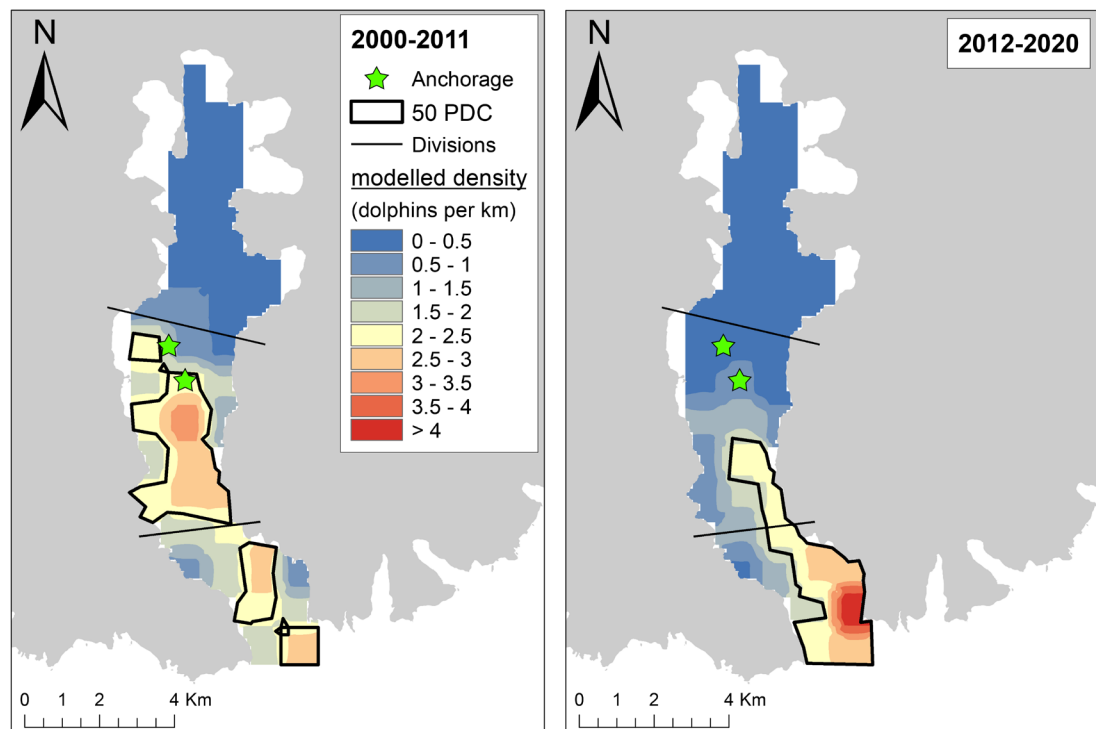


Figure 11. Kernel density estimation (KDE) of summer (December to February) Hector’s dolphin sightings at Akaroa Harbour before (2000–2011) and after (2012–2020) the four-fold increase in cruise ship visits. Figure shows the modelled density estimates for each period, 50 PDC areas (highest 50% of the KDE), main cruise ship anchorages, as well as lines demarcating the inner, middle, and outer areas of the harbour. Reproduced from Carome et al. (2022).

3.6 The effect of vessel traffic on dolphin distribution in Akaroa Harbour

There were 132 days for which the T-POD collected data successfully from 08:00 to 18:00. Of these, 111 days, all between 3 December 2019 and 3 May 2020, contained values for the complete suite of explanatory variables.

During these 111 days, there were 1,080 tour trips (627 dolphin-watching trips, 453 swim-with-dolphins trips), 41 cruise ship visits, and 1,428 non-tour motor vessel observations. There were no tour trips or cruise ship visits from 24 March to 3 May 2020 during the nationwide COVID-19 lockdown, and only 35 vessels were observed from Nine Fathom Point during this time. Dolphins were detected acoustically on the T-POD on 94% of the deployment days. See further details about the modelling in Appendix 6 and Carome et al. (2023a).

Table 4. Results of the model selection for generalised additive models on dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point, Akaroa Harbour from 3 December 2019 to 3 May 2020 (111 days including lockdown). The top model (rank 1) and next best competing model (rank 2) are shown. The table shows degrees of freedom (df), Akaike Information Criterion (AICc) score, difference in AICc score (Δ AICc), model weighting, adjusted coefficient of determination (R^2), and the percentage deviance explained (% d.e.).

RANK	MODEL	DF	AICC	Δ AICC	WEIGHT	R^2	% D.E.
1	s(Dolphin_tour) + s(Swell_height_t1) + Swell_direction	11	736.16	0	0.896	0.179	33.3%
2	s(Dolphin_tour) + s(Swell_height_t1)	9	740.46	4.30	0.104	0.130	27.8%

Table 5. Statistics for (a) parametric and (b) smoothed predictor variables included in the top generalised additive model on dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point, Akaroa Harbour from 3 December 2019 to 3 May 2020 (111 days including lockdown). We selected 'mixed' swell direction as the reference level for parametric coefficients, therefore the estimates for north and south swell reflect the difference from days with mixed swells.

(a) Parametric coefficients

	COEFFICIENT	STANDARD ERROR	Z-VALUE	P-VALUE
Intercept	2.33	0.13	18.54	$< 2 \times 10^{-16}$
Swell_direction_t1 (north)	0.39	0.41	0.95	0.34
Swell_direction_t1 (south)	-0.55	0.21	-2.57	0.01

(b) Approximate significance of smooth terms

PREDICTOR	ESTIMATED DEGREES OF FREEDOM	χ^2	P-VALUE
Dolphin_tour	2.66	20.89	0.0002
Swell_height_t1	3.58	12.88	0.0065

The best model for the period of time including lockdown included the number of dolphin tours on a given day and the swell parameters for the day before (Tables 4 and 5). There was an increasing trend in dolphin detection rate at Nine Fathom Point when there were between zero and ten dolphin tours per day, and a decreasing trend in dolphin detection when there were higher levels of dolphin tourism per day. Specifically, when there were more than twelve trips in a given day, there were fewer DPMs (minutes during which dolphins were detected) with increasing number of permitted tour trips. Swell height also had a significant effect, with fewer dolphin detections on days following larger swells, although small sample sizes restrict interpretation at very small and very large swell heights. There were also fewer dolphin detections on days following southerly swells (Fig. 12).

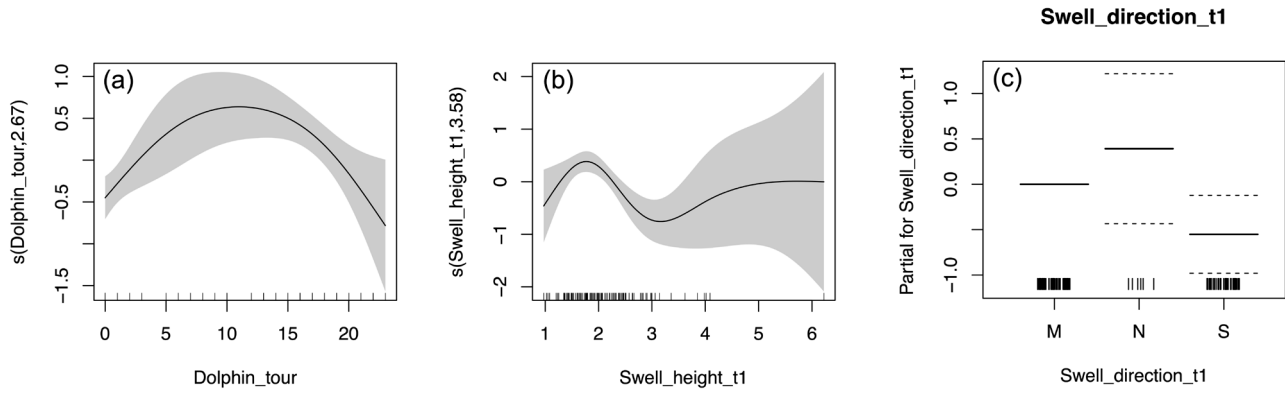


Figure 12. Predicted smoothing functions for continuous explanatory variables for (a) the number of dolphin tour trips in Akaroa Harbour, and (b) swell height on the day before. A partial dependence plot (c) shows the categorical predictor Swell_direction_t1, for the swell direction (M = mixed, N = north, S = south) on the day prior. This was included in the top-ranked generalised additive model on dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point, Akaroa Harbour, from 3 December 2019 to 3 May 2020 (over 111 days). The y-axis values are the predicted effect of each variable on the response variable, DPM per day. The 95% confidence interval of the response variable is represented by the shaded area in (a) and (b), or the area between dotted lines in (c). The y-axes vary in scale and show the estimated degrees of freedom in brackets. Reproduced from Carome et al. (2023a).

Table 6. Results of model selection for generalised additive models on the dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point, Akaroa Harbour from 3 December 2019 to 23 March 2020 (over 74 days excluding lockdown). All models within three AICc points of the top-ranked model are shown. The table shows the degrees of freedom (df), AICc score, difference in AICc score (Δ AICc), model weight, adjusted R^2 (R^2) and percent deviance explained (% d.e.).

RANK	MODEL	DF	AICC	Δ AICC	WEIGHT	R^2	% D.E.
1	s(Day_of_season) + s(Cruise) + s(Motor) + s(Swell_height_t1) + Swell_direction_t1	10	533.36	0	0.234	0.106	35.1%
2	s(Motor) + s(Swell_height_t1) + Swell_direction_t1	6	534.54	1.18	0.130	0.076	22.5%
3	s(Day_of_season) + s(Motor) + s(Swell_height_t1) + Swell_direction_t1	9	534.64	1.28	0.124	0.108	30.3%
4	s(Cruise) + s(Motor) + s(Swell_height_t1) + Swell_direction_t1	9	534.70	1.33	0.120	0.101	30.7%
5	s(Day_of_season) + s(Cruise) + s(Swell_height_t1) + Swell_direction_t1	9	535.13	1.77	0.097	0.113	30.6%

When the period of time during the nationwide lockdown was excluded, 74 days of DPM data had a complete suite of predictor variables.

The best five models were similarly well-supported by the data. The top-ranked model of DPM for the period of time excluding lockdown included: day of season, cruise ship presence, non-tour motor vessel observations at Nine Fathom Point, and swell height and direction on the day before (Tables 6 and 7).

The dolphin detection rate increased between 3 December 2019 and 23 March 2020. There was a significant decrease of DPMs during increasing non-tour motorised traffic. There were also fewer dolphin detections when one or more cruise ships were in Akaroa Harbour. Swell height and direction on the day before showed similar effects to those observed in the models including lockdown. However, when measurements during lockdown were excluded (and day of season included), the effect of each swell direction was more distinct. There were significantly more dolphin detections on days following a northerly swell and significantly fewer detections on days following a southerly swell (Fig. 13).

Table 7. Statistics for (a) parametric and (b) smoothed predictor variables included in the top generalised additive models (GAMs) on dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point, Akaroa Harbour, from 3 December 2019 to 23 March 2020 (74 days excluding lockdown).

(a) Parametric coefficients

	COEFFICIENT	STANDARD ERROR	Z-VALUE	P-VALUE
Intercept	2.56	0.14	18.10	$< 2 \times 10^{-16}$
Swell_direction_t1 (north)	0.98	0.45	2.16	0.030
Swell_direction_t1 (south)	-0.76	0.27	-2.78	0.005

(b) Approximate significance of smoothed terms

PREDICTOR	ESTIMATED DEGREES OF FREEDOM	χ^2	P-VALUE
Day_of_season	1.00	3.90	0.048
Cruise	1.00	3.72	0.054
Motor	1.16	7.21	0.021
Swell_height_t1	3.16	14.20	0.012

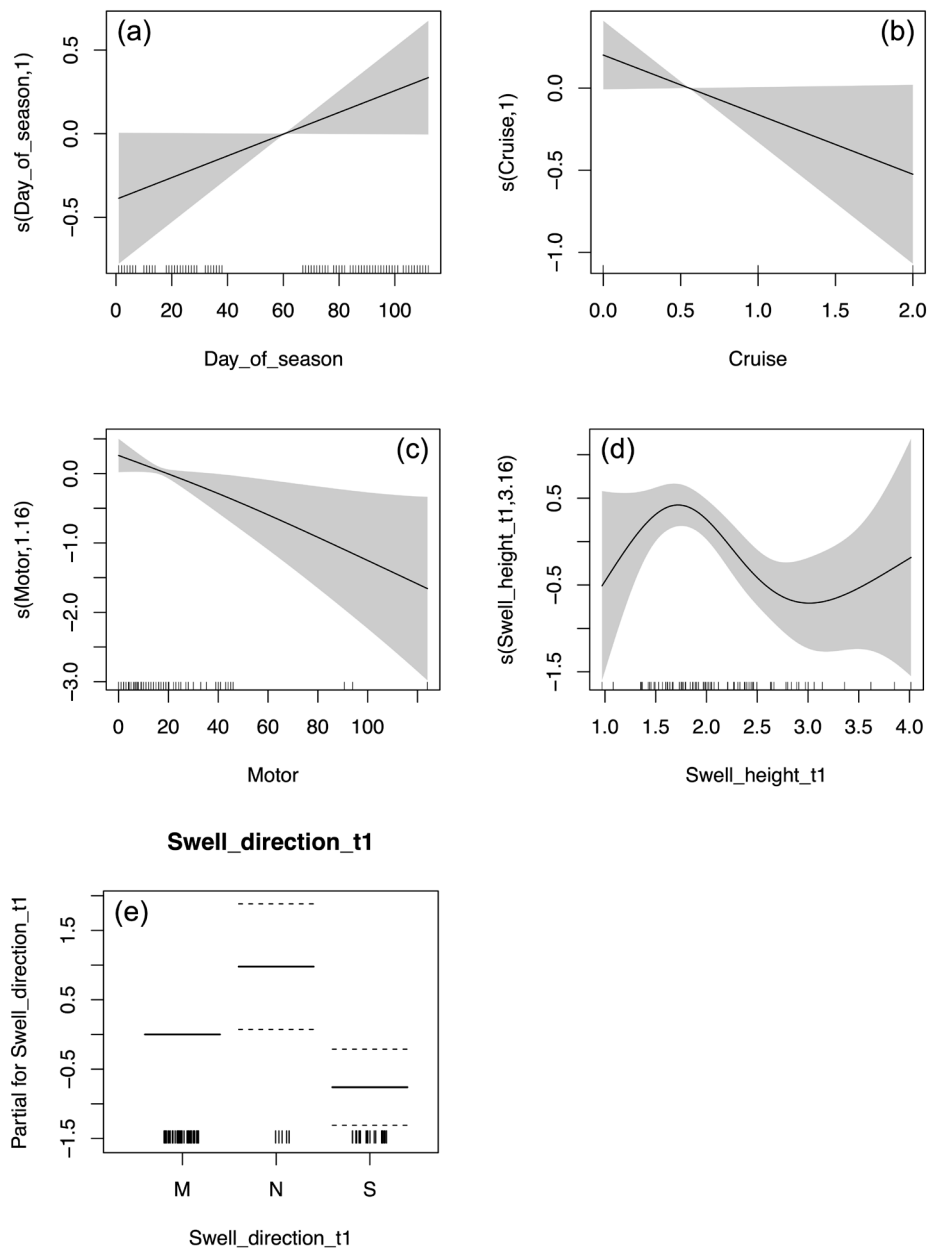


Figure 13. Predicted smoothing functions for continuous explanatory variables for (a) the day of season (3 December 2019 = day 1), (b) the number of medium or large cruise ships present in Akaroa Harbour, (c) the number of motor vessel observations at Nine Fathom Point excluding dolphin tours, and (d) the swell height on the day before. (e) A partial dependence plot for the categorical predictor of swell direction (M = mixed, N = north, S = south) on the day before, $\text{Swell_direction_t1}$. These are included in the top-ranked GAM on dolphin detection rate (DPM 08:00 to 18:00) at Nine Fathom Point from 3 December 2019 to 23 March 2020 (74 days excluding lockdown). The y-axes values are the predicted effect of each variable on DPM per day. The 95% CI of the response variable is represented by the shaded area in (a) to (d), and the area between dotted lines in (e). The y-axes show the estimated degrees of freedom in brackets. Reproduced from Carome et al. (2023a).

4. Discussion

This study draws on the long-term monitoring of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula to estimate demographic parameters and patterns of distribution. It also adds targeted research on the distribution of vessel traffic to investigate the effects of tourism.

The result is a comprehensive assessment of Hector's dolphins in one of their national strongholds, providing new insights into the factors that influence their conservation status.

4.1 Updated survival rate for 2009 to 2019

This study provided a robust estimate of apparent survival rate for Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula, the first update since 2006.

Encouragingly, the estimates of the mean apparent annual survival increased during successive management periods at Banks Peninsula in which additional protection measures to mitigate fisheries bycatch were introduced. The mean annual survival rate for the most recent management period between 2009 and 2019 was 0.923 (95% HDI 0.881–0.964).

Our results support the findings of Gormley et al. (2012) who demonstrated that survival rate of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula increased following the designation of the first protection measures in 1988 (i.e., establishing the sanctuary). Until now, however, the effectiveness of the additional protection measures introduced in 2008 had not been assessed. We demonstrate that the mean annual survival rate was greater in 73.5% of model runs following the new protection measures. This study therefore provides the first evidence that the increased protection introduced in 2008 has helped mitigate bycatch further.

Gormley et al. (2012) cautioned that this 'before and after' design without controls or replicates means that other explanations for increasing survival rate, such as changes in ocean temperatures or prey availability, cannot be ruled out. While not a true replicate of Gormley's work, our finding that survival rate likely increased following further protection measures adds weight to the argument that bycatch poses a serious threat to Hector's dolphins, and that marine protected areas can mitigate these impacts.

Recreational and tour boat traffic increased at the same time as the increase in protection measures to mitigate bycatch at Te Pātaka-o-Rākaihautū/Banks Peninsula (Stone and Yoshinaga 2000; Carome 2021). Cruise ship visits have also increased during this period (Wilson et al. 2015). It is not possible to determine whether these additional pressures have had an impact on survival rates of Hector's dolphins. However, the fact that survival rates have been increasing over the last 35 years suggests that the magnitude of impacts due to tourism do not outweigh the threats posed by fisheries.

It is not certain whether the increases in the survival rate of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula since the study began are sufficient to result in growth of the population. Gormley et al. (2012) estimated that with an annual survival rate of 0.917, the population was likely stable or slowly decreasing. The most recent estimate of 0.923 from this study suggests a similar, though slightly more optimistic, scenario.

With the mean fecundity rate of 0.205 used by Gormley et al. (2012), a survival rate of approximately 0.93 would be needed for population growth (see Fig. 5 in Gormley et al. 2012). Using more optimistic reproductive parameters in a population growth model for seals and dolphins (Reilly 1984; Reilly and Barlow 1986), Slooten and Lad (1991) estimated a survival rate of at least 0.92 was needed for a growing population. For the period 2009–2019, our

model gave a 37.9% probability apparent survival was 0.93 or above, and a 56.0% probability it was 0.92 or above.

While Gormley et al. (2012) estimated a 90% probability that the survival rate improved after the sanctuary was created and the first protection measures were implemented in 1988, we estimated a 57% probability it had increased. This difference may be driven by a relatively higher mean survival rate estimated for the pre-sanctuary period in our analysis (Table 4), and/or the relatively low annual survival rates estimated during 2005–2008 (Fig. 8).

Fewer false negatives may explain why we estimated a higher mean survival rate in the pre-sanctuary period. False negatives (when two photographs of the same individual are mistakenly identified as two unique individuals) cause a negative bias in survival rate through the creation of ‘ghost’ capture histories with lower recapture rates (Friday et al. 2008). Stricter data inclusion criteria in our analysis, as well as a recent audit of the University of Otago dolphin photograph database (involving a reassessment of photograph quality and distinctiveness of individuals), likely reduced the number of false negatives in our dataset compared to Gormley et al. (2012). For example, unlike Gormley et al. (2012), we excluded marks that differ between the right- and left-hand sides of the body (e.g., body pigmentation and scars), since these have been shown to result in false negatives (Hiby et al. 2013).

The relatively low estimates of survival rate between 2005 and 2008 may reflect real variations in survival and/or permanent emigration, or they may be an artifact of the modelling or sampling process which led to bias during these years. Notably, from 2005 to 2008 there was relatively higher survey effort on the north side of Te Pātaka-o-Rākaihautū/Banks Peninsula, where aerial surveys have suggested higher dolphin densities further offshore compared to the rest of the peninsula (MacKenzie and Clement 2014). This could bias the estimation of capture probability, potentially leading to a negative bias in estimates of survival rate. Therefore, work is currently in progress to explore models that can better account for uneven survey effort.

4.2 Reproductive output

We used the proportion of calves (up to one year old) observed in the population as a measure of fecundity of Hector’s dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula. The proportions were highly variable, ranging from less than 1% to more than 8% of dolphins sighted.

We are confident that variations in the proportions of calves observed reflect real variation in the fecundity of the population. The cause of these variations is not known, but they may be driven by environmental change or variations in abundance of prey (Mann et al. 2000; Ward et al. 2009). This warrants further investigation. There was no evidence of a trend in the proportion of calves over the 36 years of the study.

There was spatial variation in the proportion of calves, with area 5 (in Akaroa Harbour; Fig. 2) and area 6 (the south-west of Te Pātaka-o-Rākaihautū/Banks Peninsula), having higher values than area 3 (north) and area 4 (to the east). Although there is no evidence that Hector’s dolphins have ‘nursery areas’ at Te Pātaka-o-Rākaihautū/Banks Peninsula (Webster 2008), it is possible that certain areas have environmental conditions which are preferred by nursing mothers (e.g., Gibson et al. 2013). The proportion of calves observed in Akaroa Harbour did not differ from the remainder of the study area, suggesting that exposure to tourism and recreational vessels is not a significant driver of habitat selection by nursing mothers.

4.3 Hector's dolphin population size

In addition to using the photographic capture-recapture data to estimate survival rates and the reproductive outputs of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula, we also attempted to use the data to better understand the status of the population by estimating the abundance of dolphins.

Estimating abundance of animal species from this type of data can be challenging. Estimates typically have greater bias and lower precision than estimates of survival rate (Pollock et al. 1990). An added complication in this situation is the low mark rate of Hector's dolphins, which has changed during the course of our study (Wickman et al. 2021b).

We therefore attempted to mitigate heterogeneity in capture probability (to better meet our assumption that all individuals have the same probability of being captured) by estimating abundance only for the area and timeframe when the survey effort was most consistent. However, capture heterogeneity in our model likely means that our estimates of abundance are biased low (Seber 1982; Rosenberg et al. 1995; White and Cooch 2017). Since our annual estimates of capture probability are low, negative bias in our estimates of abundance are even more likely (Otis et al. 1978; White et al. 1982; Rosenberg et al. 1995, White and Cooch 2017). Therefore, it is important our estimates are interpreted as an index rather than a definitive population size (i.e., they are useful for investigating relative change within our study, but should not be used as a measure of absolute abundance for the area).

Our estimate of abundance of Hector's dolphins that used the area between Birdlings Flat and Steep Head during summer 2020 was 1,868 (95% HDI 988–3,154). Comparisons with previous estimates are complicated by different survey methods and different survey areas. The mean of our estimate is larger than the two estimates derived using data from the 1990s (897, Dawson et al. 2004; 1,119, Gormley et al. 2005), both of which applied to the entire Banks Peninsula Marine Mammal Sanctuary. Our estimate is more similar to the estimates from aerial line-transect surveys in 2013 (MacKenzie and Clement 2014). They estimated an abundance of 1,684 (95% CI: 978–2,900) dolphins over a similar, but slightly longer stretch of coast, out to four nautical miles offshore, or an abundance of 2,699 (95% CI 1386–5424) out to 20 nautical miles.

Although our estimate of Hector's dolphin abundance was larger than the estimates derived in the 1990s (Dawson et al. 2004; Gormley et al. 2005), this does not necessarily mean that population size has increased between these two periods. The estimate using line-transect methods (Dawson et al. 2004) applied only to individuals within four nautical miles of the coastline. Our capture-recapture estimate, however, applies to all individuals that used our surveyed area. Although Gormley et al. (2005) also used the capture-recapture approach, they used one boat instead of two, making our estimates difficult to compare.

We investigated potential change in the population size over time by estimating the trend in abundance of marked Hector's dolphins between 2002 and 2020, when the survey effort was relatively consistent. The slope of the regression line was positive, but not significantly different to zero. Therefore we have insufficient evidence to conclude that population size has increased over the last 20 years. However, the ability to detect population change from trends in abundance is usually poor (Taylor et al. 2007), especially for small populations (Taylor and Gerrodette 1993), and we acknowledge that the precision of our annual abundance estimates is low. Our results suggest the population is either stable or slowly increasing, but do not imply a large change in abundance over time.

4.4 Proportion of dolphins using Akaroa Harbour

Based on the proportions of marked animals encountered, we estimate that about half of the Hector's dolphins in the Te Pātaka-o-Rākaihautū/Banks Peninsula population use Akaroa Harbour at some point during their lives. This estimate was consistent when using all the available photographic identification data, or the data gathered since 2000 when surveys were more evenly distributed throughout the area.

The estimated proportions are consistent with results of previous analyses. For example, Rayment, Dawson, Slooten, Bräger et al. (2009) found that the core alongshore home-ranges of half of the 20 individuals in their study were in Akaroa Harbour, and Brough et al. (2019) demonstrated a consistent hotspot of distribution in the harbour. The dolphin tourism industry operates almost exclusively in Akaroa Harbour and in waters immediately outside the harbour entrance. Recreational vessel traffic probably exists in the harbour at higher densities than anywhere else on Te Pātaka-o-Rākaihautū/Banks Peninsula. Therefore, a relatively large proportion of the Te Pātaka-o-Rākaihautū/Banks Peninsula dolphin population is exposed to intensive tourism and recreational boating pressure.

4.5 The effect of cruise ships on distribution of Hector's dolphins in Akaroa Harbour

This study presents the first long-term analysis of species distribution in relation to cruise ship tourism. There is a clear and obvious change in the distribution of Hector's dolphins during summer at Akaroa Harbour over a 20-year period, observed through the fine-scale spatial analysis of relative density.

The distribution of dolphins in the harbour has shifted southward and contracted between 2000 and 2011, and 2012 and 2020. Before 2011 the designated anchorages for cruise ships were either within or just outside the northernmost areas of core use by dolphins. After this, the core habitat shifted to areas more than one kilometre south of the cruise ship anchorages. This shift, supported by both kernel density and sighting rate analyses, happened at the same time as a more than fourfold increase in annual cruise ship visits.

Several direct and indirect pressures from cruise ship tourism are likely to influence the habitat preferences of Hector's dolphins. Potential impacts include increased ambient noise, increased risk of vessel strike, increased exposure to dolphin tourism and cascading trophic effects from seafloor habitat degradation (Lloret et al., 2021). Ambient noise is undoubtedly increased by cruise ship presence (e.g., Frankel and Gabriele 2017):

- during arrival to and departure from anchorage (e.g. propellor cavitation, Wittekind and Schuster 2016)
- at anchorage – when generators and engines must maintain power (Akaroa Harbour Operating Requirements, 2019), and
- from tender vessels shuttling passengers to and from Akaroa Wharf for up to 11 hours per day.

For species such as cetaceans that primarily interface with the world acoustically (Au 1993; Sayigh 2014), increased ambient noise can mask communication (Richardson et al. 1995; Van Parijs and Corkeron 2001), disrupt foraging (Lusseau et al. 2009) or induce sub-lethal stress responses (Rolland et al. 2012).

In addition to increased noise, it is likely that repeated, direct damage to the seafloor environment is caused by turbulence from main engine propulsion and thrusters, and anchor chain scour from cruise ships. The soft silt seafloor around cruise ship anchorages at Akaroa (Hart et al. 2009) is frequently and substantially disturbed by vessels at anchor, creating

extensive sediment plumes. Damage to soft sediment benthos (mainly invertebrates living on the seafloor) can cause loss of structure-forming and long-lived keystone species, reducing both habitat complexity and biodiversity (Handley et al. 2014; Broad et al. 2020). Such damage may threaten ecosystem function, exerting bottom-up trophic impacts on top predators (Frank et al. 2007). Given that the habitat use of Hector's dolphins is strongly linked to the characteristics of benthic habitats (e.g., substrate type; Brough et al. 2019), it is possible that seafloor disturbance may impact them through the distribution of their prey.

The frequent influx of cruise ship passengers also resulted in an extended peak in the dolphin tourism season. Outside of the Christmas to New Year holiday season, tour operators capitalise on cruise ship visits by running additional trips to view and swim with Hector's dolphins at Akaroa (Carome 2021). Prolonged presence of vessels, not just the noise they produce, can play a substantial role in disturbance to dolphins (Pirota et al. 2015). Tour vessels constitute the majority of human interaction with Hector's dolphins (Martinez 2010). This means long-term, increased exposure to tour vessels may help to explain changes in summer distribution over time.

Other changes in Akaroa Harbour – such as fluctuations in recreational vessel traffic or oceanographic parameters – should not be overlooked. For example, Aotearoa New Zealand experienced largely positive sea surface temperature anomalies since the end of the twentieth century. The coastal Canterbury region warmed at the fastest rate of Te Waipounamu/South Island coastal regions (Pinkerton et al. 2019). Continued ocean warming is projected (Law et al. 2018). This will impact productivity, prey, and top predators (Wernberg et al. 2012; Wernberg et al. 2016; Pecl et al. 2017).

Another change in Akaroa Harbour during the study period was the establishment of the Akaroa Marine Reserve in 2014. This could have made the habitat in the outer harbour more appealing to Hector's dolphins. However, with the observed shift in dolphin distribution occurring before the introduction of the marine reserve, and the time typically taken for effects of marine reserves to become apparent (Babcock et al. 2010), it is unlikely that the reserve has been a driver of the observed contraction and shift southwards in Hector's dolphin distribution.

4.6 The effect of vessel traffic on fine-scale distribution in Akaroa Harbour

The top generalised additive models (GAMs) explaining the dolphin detection rate at Nine Fathom Point from 3 December 2019 to 3 May 2020 all contained at least one parameter related to vessel traffic. Although T-PODs cannot directly measure animal density, several cetacean studies demonstrated a correlation between animal density and the number of acoustic detections (e.g., Marques et al. 2009; Sveegaard et al. 2011; Kyhn et al. 2012; Jaramillo-Legorreta et al. 2017). While changes in detection rate could also reflect a change in acoustic behaviour of the Hector's dolphins in the area, there have been no studies to investigate this. The simplest explanation is that the density of Hector's dolphins was influenced by vessel traffic. DPM per day decreased with: increasing motor vessel traffic, increasing cruise ship presence, and at high levels of dolphin tourism. Dolphin tourism vessels were not included in counts of motor vessel traffic and were monitored separately via GPS.

In the set of models that included the period of COVID-19 lockdown, the dolphin detection rate showed a significant relationship with tour vessel traffic. Including tour vessel traffic in models required the exclusion of the day of season variable due to high concurrency, a form of correlation between predictor variables in nonlinear models. This concurrency is likely related to low detections of Hector's dolphins during the nationwide lockdown period (23 March to 14 May 2020), when tourism was non-existent. These relatively low detection rates during lockdown likely resulted from the seasonal change in distribution whereby dolphins tend

to move offshore and out of the harbour in autumn and winter (Rayment et al. 2010; Dawson et al. 2013).

There is a clear negative correlation between DPM and high levels of dolphin tourism. A negative trend in dolphin detections at Nine Fathom Point is apparent when there are more than 12 dolphin tour trips a day, with a strong negative effect at highest levels of dolphin tourism. The highest levels of commercial tourism were generally limited to the Christmas and New Year holiday period, and other days when cruise ships were present. The highest 10-day stretch of dolphin tourism during December 2019 to May 2020 occurred between 24 December and 2 January (20.8 trips per day on average). Apart from 2 January, all days with more than 20 permitted tour trips occurred when cruise ships were anchored in the harbour (Carome et al., unpub. data). While dolphin tourism only represents one facet of vessel traffic, dolphin tour vessels constitute the majority of human interactions with Hector's dolphins (Nichols et al. 2001; Martinez 2010).

When the COVID-19 lockdown was excluded from models, we observed the negative effects of cruise ship presence and significant negative effects of the increasing traffic of motor vessels (excluding dolphin tours) on dolphin presence at Nine Fathom Point. These models are made more robust, at the cost of reduced sample size, by incorporating day of season. This helped account for observed seasonality in the inshore presence of Hector's dolphin (Rayment et al. 2010; Dawson et al. 2013). As discussed previously, cruise ships present a range of potential impacts, including increased ambient noise (Frankel and Gabriele 2017) and seafloor habitat degradation (Broad et al. 2020). Additionally, cruise ships result in increased tourism pressure on Hector's dolphins as tourism operators at Akaroa town run significantly more trips on days when cruise ships are present (Carome 2021).

The observed effect of motorised vessel traffic on Hector's dolphins is more directly interpretable, as it is a significant effect of a predictor recorded at high-resolution in the same area where dolphin presence was measured. As more non-tour motor vessels entered the Nine Fathom Point study site, we detected fewer dolphins on average. These results support relationships between vessel traffic and dolphin habitat use that were described previously (e.g., Allen and Read 2000; Lusseau 2005; Rako et al. 2013; Marley et al. 2017), demonstrating a relationship between vessel traffic and dolphin presence can be observed using automatic detection methods and robust statistical modelling. It has been suggested that Hector's dolphins at Akaroa have become habituated to vessel traffic (Stone and Yoshinaga 2000), and a significant relationship between vessel traffic and dolphin presence was not observed in previous research (Nichols et al. 2001; Martinez 2010). Three immediate possibilities exist:

- Hector's dolphins at Akaroa have become sensitised (Allaby 1999) to vessel traffic during the last decade
- vessel traffic levels observed in the present study are higher than levels previously tolerated by local dolphins, or
- previous studies did not possess the statistical power to detect the effects observed in this study.

Previously described habituation (Stone and Yoshinaga 2000) may have been a misinterpretation of tolerance of human activity (using the meaning of Bejder et al. 2006, 2009). Vessel traffic may have surpassed what is tolerable for these dolphins during the summer of 2019 to 2020.

While high concurrency (smooth terms that can be approximated by others) between cruise ship, dolphin tour, and non-tour motorised traffic variables makes it difficult to truly isolate their effects, a key mechanism of action affecting dolphin presence is similar for all three.

Each of these predictors presents a source of anthropogenic noise in Akaroa Harbour, with the potential to impact Hector's dolphins through:

- masking of their own or environmental sounds (Clark et al. 2009; Erbe et al. 2016)
- temporary threshold shifts in hearing (Erbe 2002; Mooney et al. 2012), or
- displacement of prey species (Becker et al. 2013).

Cruise ships undoubtedly introduce noise to the harbour from engine noise (e.g., Arveson and Vendittis 2000) and the bubbles generated by propellers (cavitation; e.g., Wittekind and Schuster 2016) on arrival and departure. Cruise ships also contribute noise while at anchor. Generators run continuously, the propulsion engines must remain on-line (i.e. rather than shut down; Akaroa Harbour Operating Requirements 2019), and tender vessels shuttle passengers to and from the Akaroa wharf. Both tour vessels (e.g., Lusseau 2005; Constantine et al. 2004) and other motor vessels are shown to change cetacean behaviour (e.g., Van Parijs and Corkeron 2001; Buckstaff 2004; Williams et al. 2006). Displacement from the vicinity of disturbance has also been observed (Allen and Read 2000; Lusseau 2005; Bejder et al. 2006). While isolating the exact cause of these behavioural changes presents a challenge, recent research has shown that noise levels can play a key role (Sprogis et al. 2020). It is undeniable that motorised vessel activities increase ambient noise (Jensen et al. 2009; Erbe 2002 Erbe et al. 2016; Hermanssen et al. 2019), and therefore have potential to disrupt, disturb and displace Hector's dolphins.

5. Recommendations for managing the impacts on Hector's dolphins

The results of this study have significant implications for managing the impact of tourism on Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula. We showed that a change in distribution of dolphins coincided with the dramatic increase in use of Akaroa Harbour by cruise ships. In our opinion, cruise ship tourism in the harbour should therefore be limited to the levels that existed prior to the Christchurch earthquakes in 2011. Decisions to increase cruise ship visits beyond those levels should only be made when the mechanisms of the impact on the dolphins are clear, and once acceptable thresholds of impact are established.

Furthermore, given the range of evidence suggesting Hector's dolphins are displaced from core habitat by vessel traffic, we recommend using immediately actionable tools to reduce vessel pressure on these dolphins. Suitable tools include limits on maximum daily number of dolphin tour trips, as well as speed limits and entry restrictions in core dolphin habitat.

Finally, as stated in New Zealand's Marine Mammal Protection Regulations 1992 (section 6), in order to meet the definition of ecotourism, and the minimum requirements to receive a marine mammal viewing permit, tourist operations should be required to demonstrate a net conservation benefit to the Hector's dolphin population on which they are focused. Therefore, organisations managing these impacts on dolphins should assess the educational and conservation aspects of dolphin tours.

The evidence presented here suggests that cumulative human pressures on Hector's dolphins in Akaroa Harbour were unsustainable before the COVID-19 pandemic. There is now an opportunity to reframe how Akaroa Harbour is used, and an obligation to ensure that this use does not have an adverse impact on Hector's dolphins and other species.

Delays in response elsewhere to the evidence of impacts from tourism – such as in Tokerau/Bay of Islands (Constantine 2001; Constantine et al. 2004; Peters and Stockin 2016) – had significant consequences (Fumagalli et al. 2021). With the high likelihood of impact suggested here, the burden of proof should be placed on the cruise ship and tourism industries to provide evidence of sustainability (Constantine and Bejder 2008). Until that sustainability can be demonstrated, we recommend that the moratorium on issuing dolphin watching permits be continued beyond the next review in 2026.

Management of tourism should aim to support the Hector's dolphin population's use of all of its available habitat. Therefore, we recommend restrictions in both current core-use areas of the dolphins (e.g., the outer part of the Akaroa Harbour) and the core areas in use before the 2011 earthquakes (e.g., mid harbour). Tourism restrictions should be monitored for effectiveness using further research on dolphin distribution, with return of dolphin density to historic core-use areas a potential marker for successful management.

In summary, we recommend:

- dolphin tourism should be capped at a maximum of 20 trips per day across all operations in Akaroa Harbour, and preferably at 12 trips per day
- cruise ship visits should be limited to the levels before the 2011 earthquakes
- area-based restrictions should be imposed to reduce vessel pressure on dolphins
- the educational and conservation aspects of dolphin tours should be assessed
- the moratorium on dolphin-focused permits should remain in place beyond 2026.

6. Areas of future research

Our study revealed additional research questions which could further clarify the effects of tourism on Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula.

What are the most likely mechanisms explaining the change in distribution of Hector's dolphins in Akaroa Harbour which coincided with the increase in cruise ship tourism?

This could be addressed by quantifying noise levels in the presence and absence of cruise ships, targeted investigations of benthic disturbance at cruise ship anchorages, and measuring spatial variation in benthic species diversity and the prey of Hector's dolphins. Data will need to be gathered over several years if these effects are to be teased apart. The response in dolphin distribution to the reduction in cruise ship visits during and since the COVID-19 pandemic should also be investigated. Continuation of the long-term data collection on Hector's dolphins at Akaroa Harbour is of highest priority. Data from these surveys has the power to elucidate whether predominant impacts from cruise ships are exerted in the short-term (e.g., noise) or require years to resolve (e.g., seafloor degradation).

What is the effect of the establishment of the Akaroa Marine Reserve on distribution of Hector's dolphins?

Long-term monitoring data could be used to compare sighting rates within and near Akaroa Marine Reserve before and after the establishment of the reserve in 2014, with reference to suitable control areas. Observing Hector's dolphins in Pohatu Marine Reserve – established in Pōhatu/Flea Bay in 1999 and therefore well before the major expansion of cruise ship tourism in Akaroa Harbour – may help to disentangle the simultaneous effects of multiple drivers of dolphin distribution.

What is the direction of displacement of dolphins in response to vessel traffic at Nine Fathom Point? Are relationships between human activities and distribution of dolphins similar throughout the harbour, and what role is played by variations in environmental conditions?

Additional passive acoustic monitoring locations could be used to determine whether dolphins are displaced into, or out of, Akaroa Harbour. Further oceanographic and prey sampling could extend models beyond where dolphins are found, to why they prefer certain areas.

What are the drivers of variability in demographic parameters of Hector's dolphins?

Long-term monitoring data could be used to investigate whether temporal variations in fecundity and apparent survival rate are related to changing oceanographic conditions and/or variations in human activities (other than protection from bycatch). The analyses could include methods to estimate fecundity more effectively, for example by trialling non-linear effects and accounting for the imperfect detection of reproductive females (e.g. Cheney et al. 2019).

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Appendix 1

Analysis methods for survival rate

In this appendix we provide details of the statistical analysis to determine how the apparent survival rate of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula varied over time, including after the establishment of the Banks Peninsula Marine Mammal Sanctuary.

Following the approach used by Gormley et al. (2012), we used a modified version of the Cormack-Jolly-Seber (CJS) model (Cormack 1964; Jolly 1965; Seber 1965) to estimate yearly apparent survival rates of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula. The CJS model allows for imperfect detection of individuals per year, and for an open population (i.e. births, deaths, immigration and emigration can occur between study periods). The Bayesian capture-recapture model used in this survival analysis was identical to that in Gormley et al. (2012), but with a third mean-survival period (post-2008), a modification to prior specification on μ_ϕ (mean probability of survival, on the logit scale), and including capture-recapture data up to 2020.

Since CJS models assume that no marks are lost and that no individuals are identified incorrectly (Pollock et al. 1990), we only included individuals with the most distinct, permanent marks (category 1 and 2 in the University of Otago catalogue). Since the inclusion of individuals sighted outside our regular survey area could introduce bias, we only included sightings between Birdlings Flat (area 30 in Fig. 2), and Sumner Head (area 16 in Fig. 2). We included only summer months (November through to March) to account for the assumption of closure within each period (Pollock et al. 1990) while also allowing for sufficient recapture data.

Bayesian hierarchical capture-recapture models separate the 'observation' process (whether or not an individual is photographed that year) from the 'state' process (whether an individual is alive or dead). First, the state of each individual is modelled, then whether it is observed, conditional on it being alive. We describe the state and observation portions of the model separately below.

State process

A Bernoulli distribution modelled individual-specific survival from the occasion of their first capture as a random variate with the states 'alive' ($A_{i,t} = 1$) or 'dead' ($A_{i,t} = 0$):

$$A_{i,t} \sim \text{Bern}(\phi_{t-1} \cdot A_{i,t-1}), i = 1, \dots, n; t = a_i + 1, \dots, T$$

where ϕ_t is the probability of survival from t to $t + 1$, n is the total number of individuals, a_i is the first year captured, and T is the total number of sampling periods. Therefore, the alive state ($A_{i,t} = 0$ or $A_{i,t} = 1$) is determined by the probability of survival in the previous year, given the individual was alive the previous year.

We used the logit scale to model survival, allowing for three different means to represent three different management periods. Our management periods were:

- 1986–1989 (no management in place)
- 1990–2008 (following establishment of the Banks Peninsula Marine Mammal Sanctuary), and
- 2009–2019 (following establishment of set-net and trawling restrictions north and south of Te Pātaka-o-Rākaihautū/Banks Peninsula).

Annual variation for each mean was treated as a random effect.

$$\text{logit} \left(\phi_{t-1} \right) = \begin{cases} \mu_{(\phi)1986-1989} + \varepsilon_{(\phi)_{t-1}}, & t = 2, \dots, 5 \\ \mu_{(\phi)1990-2008} + \varepsilon_{(\phi)_{t-1}}, & t = 6, \dots, 21 \\ \mu_{(\phi)2009-2019} + \varepsilon_{(\phi)_{t-1}}, & t = 22, \dots, T \end{cases}$$

where $\mu_{\phi 1986-1989}$, $\mu_{\phi 1990-2008}$, and $\mu_{\phi 2009-2019}$ are the means for each management period, and $\varepsilon_{(\phi)_{t-1}}$ is the random effect of survival for each year, modelled by a normal distribution:

$$\varepsilon_{(\phi)_{t-1}} \sim N\left(0, \sigma_{\phi}^2\right)$$

where σ_{ϕ}^2 is the annual process variation of survival.

Observation process

During each sampling period, each animal was ‘captured’ (a photo of suitable quality was taken, $x_{i,t} = 1$) or ‘not captured’ (no suitable photo was taken, $x_{i,t} = 0$). This observation state was modelled as the random variate of a Bernoulli distribution:

$$x_{i,t} \sim \text{Bern}(A_{i,t} \cdot p_{i,t}), i = 1, \dots, n; t = a_{i+1}, \dots, T$$

where $p_{i,t}$ is the annual probability of capture for each individual. We accounted for heterogeneous capture probabilities by using the number of times an individual was captured in the previous year as a covariate for whether or not $x_{i,t} = 1$ in the next year (Fletcher 1994, Gormley et al. 2012).

$$\text{logit}(p_{i,t}) = \alpha_t + \beta_t \cdot z_{(\text{std})i,t-1}, i = 1, \dots, n; t = 2, \dots, T$$

where α_t is the intercept of the regression, β_t is the slope of the regression, and $z_{(\text{std})i,t-1}$ is $z_{i,t-1}$ (the number of times an individual was seen in the previous year) standardised as:

$$z_{(\text{std})i,t-1} = \frac{z_{i,t-1} - 5}{10}, i = 1, \dots, n; t = 2, \dots, T$$

The intercept (α_t) and slope (β_t) were both modelled as normally distributed random effects:

$$\alpha_t = \mu_{\alpha} + \varepsilon_{(\alpha)t}, t = 2, \dots, T$$

$$\beta_t = \mu_{\beta} + \varepsilon_{(\beta)t}, t = 2, \dots, T$$

where μ_{α} and μ_{β} are the mean of α and β (respectively), and $\varepsilon_{(\alpha)t}$ and $\varepsilon_{(\beta)t}$ are the random effects of α and β (respectively). Each random effect was normally distributed:

$$\varepsilon_{(\alpha)t} \sim N\left(0, \sigma_{\alpha}^2\right)$$

$$\varepsilon_{(\beta)t} \sim N\left(0, \sigma_{\beta}^2\right)$$

where σ_{β}^2 and σ_{α}^2 were the annual process variation for α and β .

We estimated the difference in mean survival, on the logit scale, between each management period by subtracting the mean survival in the earlier period(s) from the mean survival in the later periods, as follows:

$$\Delta\mu_{\phi 2,1} = \mu_{\phi 1990-2008(2)} - \mu_{\phi 1986-1989(1)}$$

$$\Delta\mu_{\phi 3,2} = \mu_{\phi 2009-2019(3)} - \mu_{\phi 1990-2008(2)}$$

$$\Delta\mu_{\phi 3,1} = \mu_{\phi 2009-2019(3)} - \mu_{\phi 1986-1989(1)}$$

The percentage of model runs where the difference was above zero was taken as the probability that the survival of the Hector’s dolphin had increased in each of the three scenarios.

Priors and parameter estimation

We ran the model using JAGS 4.3.0 (Plummer 2003) in R 4.1.1 (R Core Team 2021) using the packages *runjags* (Denwood 2016) and *rjags* (Plummer 2022). We used a slightly informative prior on the mean estimate of survival rate (μ_ϕ) for each period to avoid unrealistic survival rates, *Logistic(2.2, 1)*, which translates to a 99% range of -3.11 to 9.92 (when transformed from the logit scale to probability, this is 0.04428 to 0.99995). To avoid the posterior distribution for annual survival rate tending towards one without evidence from our data, we used a scaled gamma distribution for the prior on σ_ϕ^2 , (*Scaled.Gamma(0.2, 2)*).

Vague prior distributions were chosen for all other parameters used to estimate capture probability ($P_{i,t}$): normal ($N(0, 0.001)$) for the mean coefficient parameters (μ_α and μ_β), and uniform (*Unif(0, 100)*) for all the variance parameters (σ_α and σ_β).

We ran four different Markov chains, each with separate initial values. To allow convergence of each chain, the first 15,000 steps were discarded, and the model was run a further 40,000 steps. To help reduce autocorrelation, each chain was thinned by every third iteration. Thus, 10,000 steps were kept from each of the four chains, giving a total of 40,000 steps that were used to give the posterior sample for each parameter. Convergence was checked visually and using the Gelman-Rubin statistic (chains are considered “converged” if $Rc < 1.1$).

Estimates of each parameter were taken as the median of their posterior distribution. Variance of each parameter was estimated from its posterior samples, with the 95% HDI (highest density interval) representing where 95% of samples in the posterior distribution lie. The posterior samples of mean survival on the logit scale were transformed to probability:

$$\phi_m = \text{expit}(\mu_{\phi_m}) = \frac{e^{\mu_{\phi_m}}}{1 + e^{\mu_{\phi_m}}}$$

where m is the management period (1986–1989, 1990–2008 or 2009–2019). The mean apparent survivals of each period ($\mu_{\phi_{1986-1989}}$, $\mu_{\phi_{1990-2008}}$ and $\mu_{\phi_{2009-2019}}$) are therefore presented in the results as $\phi_{1986-1989}$, $\phi_{1990-2008}$ and $\phi_{2009-2019}$, respectively.

Appendix 2

Analysis of reproductive output

Here we provide details of the statistical analysis to determine:

- how the reproductive rate of Hector's dolphins varied over time
- if the reproductive output of dolphins that use Akaroa Harbour differs from the population in the rest of Te Pātaka-o-Rākaihautū/Banks Peninsula.

We investigated the temporal and spatial variations in reproductive output of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula by calculating the proportion of calves (up to 1 year old) sighted in the population (e.g., Wells and Scott 1990; Bearzi et al. 1997). Proportions were calculated for each survey year, i , and each survey area, j (see Fig. 2) as follows:

$$P_{ij} = \frac{C_{ij}}{T_{ij}}$$

where, C_{ij} is the number of calves seen in year i in area j , and T_{ij} is the total number of dolphins seen in year i in area j . To be confident that young of the year were identified correctly, we only used data from the summer season, i.e., November to March. Each season is labelled with the later year, i.e., the summer season November 1998 to March 1999 is labelled as 1999. To reduce the risk of bias due to small sample sizes, we only included years and regions for which at least 100 dolphins were sighted. This effectively excluded areas 2 and 7 (see Fig. 2) due to the much lower survey effort outside Te Pātaka-o-Rākaihautū/Banks Peninsula.

We fitted a linear model to investigate if there has been a trend in proportion of calves over time. To investigate spatial variation in reproductive output we then fitted a linear model with survey region and year as explanatory variables. Lastly, to test if reproductive output in Akaroa Harbour differed from the rest of Te Pātaka-o-Rākaihautū/Banks Peninsula combined we used a paired t-test, with the proportions paired by year. Diagnostic plots of residuals revealed no concerning patterns for any of the models (Figs. A6.1, A6.2 and A6.3).

Appendix 3

Analysis of population size

In this section, we provide details of the statistical analysis to determine whether the population size of Hector's dolphins at Te Pātaka-o-Rākaihautū/Banks Peninsula varied over time.

We estimated dolphin abundance using a Bayesian hierarchical CJS model with time-varying capture probability (p_t) and different mean survival between the management periods defined above (with annual variation as a random effect). Unlike Jolly-Seber models of abundance, which use the full capture history (i.e., the zeroes before first capture) and estimate the number of individuals that are entering the population each year (even if they have not yet been captured), our model bases its abundance estimates off recaptures only.

Compared to estimates of survival rate, estimates of abundance are more sensitive to capture heterogeneity (Nichols et al. 1984; Williams et al. 2002). To minimise capture heterogeneity due to different levels of sampling in different areas each year, we restricted data to the portion of Te Pātaka-o-Rākaihautū/Banks Peninsula most regularly surveyed, between Birdlings Flat (section 30, Fig. 2) and Steep Head (section 22, Fig. 2). We also restricted the dataset to 2001–2020, after the addition of digital photography and the regular practice of using two survey vessels simultaneously during the field season. These changes could potentially lead to an increase in the proportion of the population available for capture (individuals which were unavailable for capture using the potentially less intensive, previous methodology become available for capture). Since this could make the earlier (pre-2001) and later periods (post-2001) of the study difficult to compare, we have chosen to exclusively model data from 2001–2020. As with our analysis of survival rates, we used capture-recapture data collected only during the summer months (November through to March), to account for the assumption of closure within each period (Pollock et al. 1990), while also allowing for sufficient data for the analysis.

As with the model for survival rate, the model for abundance includes:

- a state process (whether or not an individual is alive or dead), and
- an observation process (whether it is seen and photographed).

Each of these portions of the model is described separately below.

State process

The state process of the abundance model was identical to the state process for estimating for survival, but with just two management periods, since we only included data after 2001 for abundance estimation. A Bernoulli distribution modelled each individual as a random variate with the states “alive” ($A_{i,t} = 1$) or “dead” ($A_{i,t} = 0$):

$$A_{i,t} \sim \text{Bern}(\phi_{t-1} \cdot A_{i,t-1}), i = 1, \dots, n; t = a_i + 1, \dots, T$$

where ϕ_t is the probability of survival from t to $t + 1$, n is the total number of individuals, a_i is the first year captured, and T is the total number of sampling periods (20 between 2001 and 2020).

We estimated separate mean survival over two management periods:

- 2001–2008 (following establishment of the Banks Peninsula Marine Mammal Sanctuary), and
- 2009–2019 (following establishment of set net and trawling restrictions north and south of Te Pātaka-o-Rākaihautū/Banks Peninsula).

Annual variation for each mean was treated as a random effect.

$$\text{logit}(\phi_{t-1}) = \begin{cases} \mu_{\phi 2001-2008} + \varepsilon_{(\phi)t-1} & , t = 2, \dots, 9 \\ \mu_{\phi 2009-2019} + \varepsilon_{(\phi)t-1} & , t = 10, \dots, T \end{cases}$$

where $\mu_{\phi 2001-2008}$ and $\mu_{\phi 2009-2019}$ are the means for each management period on the logit scale, and $\varepsilon_{(\phi)t-1}$ is the random effect of survival for each year, modelled by a normal distribution:

$$\varepsilon_{(\phi)t-1} \sim N(0, \sigma_{\phi}^2)$$

where σ_{ϕ}^2 is the annual process variation of survival.

Observation process

To allow derivation of abundance, we only allowed capture probability (p_t) to vary by year (individual heterogeneity was not modelled). During each sampling period, each animal was ‘captured’ (a photo of suitable quality was taken, $x_{i,t} = 1$) or ‘not captured’ (no suitable photo was taken, $x_{i,t} = 0$). This observation state was modelled as the random variate of a Bernoulli distribution:

$$x_{i,t} \sim \text{Bern}(A_{i,t} \cdot p_t), i = 1, \dots, n; t = a_{i+1}, \dots, T$$

where p_t is the annual probability of capture.

Mark rate estimation

We used a Bayesian hierarchical model for estimating mark rate described by Eguchi (2014). This method was found by Wickman et al. (2021a) to be the most reliable method for accurately estimating the mark rate and its variance. To estimate the mark rate over a given period, the model requires data on the number of images with marks for each group (K_g) and the total number of images acquired for each group (T_g). Data inputs for K_g and T_g came from random photography of dolphin groups during the 2016 field season.

Field methods for estimating the mark rate in 2016 are described in Wickman et al. (2021a,b). Briefly, photos of each dolphin group were taken ‘randomly’ (dolphins were photographed whether or not they were obviously marked). If group size or composition changed substantially during a photography session, or if the photographer did not think that sufficient photographic coverage (one to three times as many photos as the apparent group size) was achieved, that group was not included in the estimation of mark rate.

The number of images with marks for each group (K_g) was modelled with a binomial distribution:

$$K_g \sim \text{Bin}(\theta_g, T_g)$$

where θ_g is the mark rate of each group. Thus, the mark rate of each group (θ_g) gives the probability of obtaining K_g images of marked individuals out of T_g total images. The parameter θ_g is drawn from a beta distribution:

$$\theta_g \sim \text{Beta}(\alpha, \beta)$$

where α and β are the shape parameters of the beta distribution. Since this beta distribution describes the distribution of group-specific mark rates (θ_g), the population-level mark rate ($\bar{\theta}_{2016}$) is taken as the mean of this distribution:

$$\bar{\theta}_{2016} = \frac{\alpha}{\alpha + \beta}$$

Abundance derivation

Abundance of the number of marked individuals was derived by dividing the number of individuals observed each year by the capture probability of that year (Seber 1982; Williams et al. 2002; Gormley et al. 2005).

$$N_t = \frac{\sum_{i=1}^n}{p_t}, t = 2, \dots, T$$

where N_t is the estimated abundance of marked individuals in year t . We then derived total abundance (for 2016–2020) by accounting for the number of unmarked individuals in the population using our estimate of mark rate ($\bar{\theta}_{2016}$):

$$N_{total_t} = \frac{N_t}{\bar{\theta}_{2016}}, t = 16, \dots, T,$$

where N_{total_t} is the total abundance. Since we estimated mark rate within our hierarchical model, the posterior distribution of N_{total_t} includes variation due to both N_t and $\bar{\theta}_{2016}$.

Past research showed that the mark rate has changed over the course of the study (likely due to increasing fishing restrictions; Wickman et al. 2021b). Since our only estimate of the mark rate after 2001 was in 2016, we estimated total population size for the period 2016–2020 only. Since fishing restrictions have not changed between 2016 and 2020, we expect very little change in mark rate over this period.

Priors and parameter estimation

We ran the model using JAGS 4.3.0 (Plummer 2003) in R 4.1.1 (R Core Team 2021) using the packages *runjags* (Denwood 2016) and *rjags* (Plummer 2022). We used a slightly informative prior on the mean estimate of survival rate (μ_ϕ) for each period to avoid unrealistic survival rates (*Logistic(2.2, 1)*), as in the methods for survival rate). To avoid the posterior distribution for annual survival rate tending towards 1 without evidence from our data, we used a scaled gamma distribution for the prior on σ_ϕ^2 (*Scaled.Gamma(2.2, 1)*). A vague prior was used for capture probability ($p_t \sim \text{Beta}(1, 1)$) and for estimating the α and β parameters in the mark rate portion of the model (*Uniform(0.001, 100)*).

We ran four different Markov chains, each with separate initial values. To allow convergence of each chain, the first 15,000 steps were discarded, and the model was run a further 40,000 steps. To help reduce autocorrelation, each chain was thinned by every third iteration. Thus, 10,000 steps were kept from each of the four chains, giving a total of 40,000 steps that were used to give the posterior sample for each parameter. Convergence was checked visually and using the Gelman-Rubin statistic (chains are considered ‘converged’ if $Rc < 1.1$).

Estimates of each parameter were taken as the median of their posterior distribution. Variance of each parameter was estimated from its posterior samples, with the 95% HDI (highest density interval) representing where 95% of samples in the posterior distribution lie.

Finally, we fitted a linear model to the yearly estimates to investigate whether the abundance of marked individuals had changed between 2001 and 2020. Diagnostic plots are shown in Fig. A6.4.

Appendix 4

Proportion of Hector's dolphin population using Akaroa Harbour

There has been considerable spatial variation in survey effort during the 36 years of monitoring at Te Pātaka-o-Rākaihautū/Banks Peninsula. For example, up until 2000, the vast majority of effort took place between Birdlings Flat and Steep Head (Fig. 2), while since then a greater proportion of effort has occurred on the northern side of the peninsula.

Furthermore, the mark rate of Hector's dolphins has decreased over time, potentially due to a decrease in encounters with fishing gear (Wickman et al. 2021b). Using capture-recapture analysis methods, it is therefore challenging to assess the proportion of dolphins around Te Pātaka-o-Rākaihautū/Banks Peninsula that use the Akaroa Harbour. We opted instead for a simpler approach, whereby the proportion of dolphins using the harbour P_{AH} was estimated as follows:

$$P_{AH} = \frac{N_{AH}}{N_{Total}}$$

Where, N_{AH} is the number of dolphins with category 1 or 2 marks which have been sighted in the Akaroa Harbour, and N_{Total} is the number of dolphins with category 1 or 2 marks in the University of Otago catalogue of dolphin photographs. In order to control for the possible effects of variable survey effort and changing mark rate we estimated this proportion for the entire period of the study, and for the period since 2000. These estimates rely on the assumption that the marked dolphins in the population are an unbiased sample of the population at Te Pātaka-o-Rākaihautū/Banks Peninsula.

Appendix 5

Effect of cruise ships on dolphin distribution in Akaroa Harbour

Fine-scale variation in the use of Akaroa Harbour by Hector's dolphins was visualised with kernel density estimation (KDE; Worton 1989) using effort weighted dolphin sightings. Separate KDEs were constructed representing the time before (2000–2011) and after (2012–2020) the four-fold increase in number of cruise ships anchoring in the middle of Akaroa Harbour. Full methods, including a description of an analysis with finer-scale temporal resolution, are described in Carome et al. (2022).

The entire harbour was not covered in all surveys. To account for uneven distribution of survey effort, the harbour was divided into 1 km² grid cells, and the proportion of total effort determined for each cell. GPS tracks of surveys were plotted in the GIS software ArcMap v 10.6.1 using the projected NZGD 2000 New Zealand Transverse Mercator coordinate system. We calculated the total survey effort (in km) in each cell for each time period using the highest GPS track resolution available for each survey. Grid cells in which total search effort was less than 10 km for the entire study period were excluded from analysis to avoid spurious effects due to sampling error. To account for seasonal variation in distribution of dolphins, we only used surveys between December and February each summer.

Weights for sightings of dolphin groups were then calculated by:

$$w_s = \frac{g_s}{D_t}$$

where g_s is the group size for sighting s and D_t was the 'on effort' distance travelled in that sighting's grid cell during the time period of the sighting. Separate calculations of sighting weight were undertaken for each time period.

The tool set Home Range Tools (MacLeod 2013) in ArcMap v 10.6.1 was used to create a fixed KDE 'with barriers' to exclude land. A grid size of 0.01 km² was defined for KDE analysis, as used by Brough et al. (2019) for Hector's dolphins. The smoothing parameter was chosen using the *ad hoc* method (Kie 2013), which involves trial and error to determine the smallest bandwidth value that produces a single continuous 95% kernel density estimate (95% KDE) contour. An optimal bandwidth value of 900 m was determined for KDEs examining the periods before and after the increase in cruise ship visits.

The 95% KDE and 50% KDE represent the minimum area in which 95% and 50% of weighted sightings occur, respectively. The 50% KDE is widely used in studies of marine mammal distribution to define core use areas or 'hotspots' (e.g., Gill et al. 2011; Bauer et al. 2015; Brough et al. 2019) and was extracted from each KDE for further analysis.

To quantify potential changes in distribution of Hector's dolphins within Akaroa Harbour over time, two analyses were undertaken:

- KDEs and core use areas were visually compared to examine potential changes in habitat preference, and
- the percentage composition of 50% KDEs with respect to the middle and outer harbour were calculated to quantify the directionality of potential shifts in habitat preference between the two time periods.

Using the inner, middle and outer harbour boundaries within Akaroa Harbour from Dawson (1991; Fig. 8), we determined the percent composition of the 50% KDE for the period before, and the period after, the increase in cruise ship visits. The percent composition metric allows for a quantification of potential shifts in relative dolphin distribution within the harbour and was calculated as:

$$\%composition = \frac{50\%KDE_{t,IMO}}{50\%KDE_t} \times 100$$

where $50\%KDE_{t,IMO}$ is the area of overlap between the core use area for time period t and the inner, middle, or outer harbour, and $50\%KDE_t$ is the core use area for time period t .

Appendix 6

Effect of vessel traffic on fine-scale distribution of Hector’s dolphins in Akaroa Harbour

We constructed generalised additive models (GAMs; Hastie and Tibshirani 1990) with a negative binomial response, in R (v. 3.6.3, R Core Team 2021) using the package *mgcv* (Wood 2020). We used these to examine the potential influence of vessel parameters on acoustic detections of dolphins at Nine Fathom Point. Only days containing a complete set of response variables were included for analyses, as GAMs cannot handle missing variables (Wood 2017). The detailed methodology and appendices can be found in Carome et al. (2023).

In addition to the vessel parameters (Table A6.1), we included environmental and temporal parameters which may have influenced the occurrence of Hector’s dolphins at the study site. Swell height and direction are shown to influence detection rates of Hector’s dolphin in Akaroa Harbour with a one-day time lag (Dittmann et al. 2016). Thus, swell parameters on the day before acoustic monitoring of dolphins were included in model construction.

Data on swell height and direction were obtained from a wave buoy (maintained by Environment Canterbury Kaunihera Taiao Waitaha, ECan) located about 17 km east of Te Pātaka-o-Rākaihautū/Banks Peninsula. From these data, mean significant wave height (H_{m0}) and predominant swell direction, in degrees, were calculated for each day (i.e., between 00:00 and 23:59). The mouth of Akaroa Harbour faces south-southeast at c. 160° true. Therefore, predominant swell direction was categorised as north (250° to 70°), south (70° to 250°), or mixed if swells from both directions were recorded on a given day.

The distribution of Hector’s dolphin varies seasonally (Rayment et al. 2010), and numbers within Akaroa Harbour peak during summer (Dawson et al. 2013). It is likely that this variation in distribution of dolphins is driven primarily by seasonal changes in distribution of prey species (Rayment et al. 2010; Dawson et al. 2013; Miller et al. 2013; Brough et al. 2020). In an attempt to account for seasonal variation in prey dynamics that may influence the distribution of Hector’s dolphins, we included the ordinal explanatory variable ‘day of season’ in model construction.

Table A6.1. List of explanatory variables offered to the generalised additive models of detection positive minutes (DPM) per day from 08:00 to 18:00 at Nine Fathom Point, Akaroa Harbour.

VARIABLE	ABBREVIATION	TYPE	DESCRIPTION
Day of season		Continuous	Ordinal day of season (starting on 3 December 2019)
Dolphin tour trips	Dolphin tour	Continuous	Number of dolphin tour trips
Medium to large cruise ships	Cruise	Continuous	Number of cruise ships anchored at Akaroa with > 500 passengers
Motor vessels	Motor	Continuous	Number of observations of motor vessels (excluding dolphin tours) off Nine Fathom Point
Non-motor/sailing vessels	Nonmotor	Continuous	Number of observations of non-motor vessels (excluding dolphin tours) off Nine Fathom Point
Swell height on the day before	Swell_height_t1	Continuous	Mean H_{m0} (00:00–23:59) at ECan wave buoy
Swell direction on the day before	Swell_direction_t1	Factor	Predominant swell direction: north (N), south (S), or mixed (M), measured at ECan wave buoy

For each model suite, an initial model containing all possible predictor variables (Table A6.1), was developed and tested for concurvity, a measure describing non-linear correlation between predictor variables (Ramsay et al. 2003). When two predictor variables had concurvity values > 0.3 , univariate models were fit for each predictor, and only the predictor producing the model with the lowest Akaike's Information Criterion accounting for small sample sizes (AICc; Akaike 1973) was retained (He et al. 2006). Day of season was shown to have unacceptably high concurvity with vessel traffic variables, forcing at least one of these variables to be excluded during model selection. In this case, the univariate model containing dolphin tour trips had a lower AICc value than the univariate model containing day of season. Thus, day of season was excluded to reduce concurvity to acceptable levels.

Concurvity was also shown to be reduced to acceptable levels by excluding the COVID-19 lockdown period (23 March to 13 May 2020), during which vessel traffic was essentially non-existent. Excluding lockdown allowed for a more complete suite of variables to be examined at the cost of reduced sample sizes, potentially weakening power to detect effects of explanatory variables. Thus, models for two time periods were generated:

- 3 December 2019 to 3 May 2020 (including the COVID-19 lockdown), and
- 3 December 2019 to 23 March 2020 (excluding the COVID-19 lockdown).

Continuous explanatory variables were smoothed by thin-plate regression splines and limited to a maximum of five basis dimensions (k) to reduce the risk of overfitting. The variable day of season was limited to three basis dimensions as overfitting was apparent at higher k -values. The function dredge (package *MuMIn*, Barton 2020) was used to rank all possible models by AICc using all combinations of the explanatory variables. Diagnostics for the top models were visually verified using the outputs of mgcv's gam.check function (Fig. A6.5, A6.6). Temporal autocorrelation was tested using the acf function in the package *stats*.

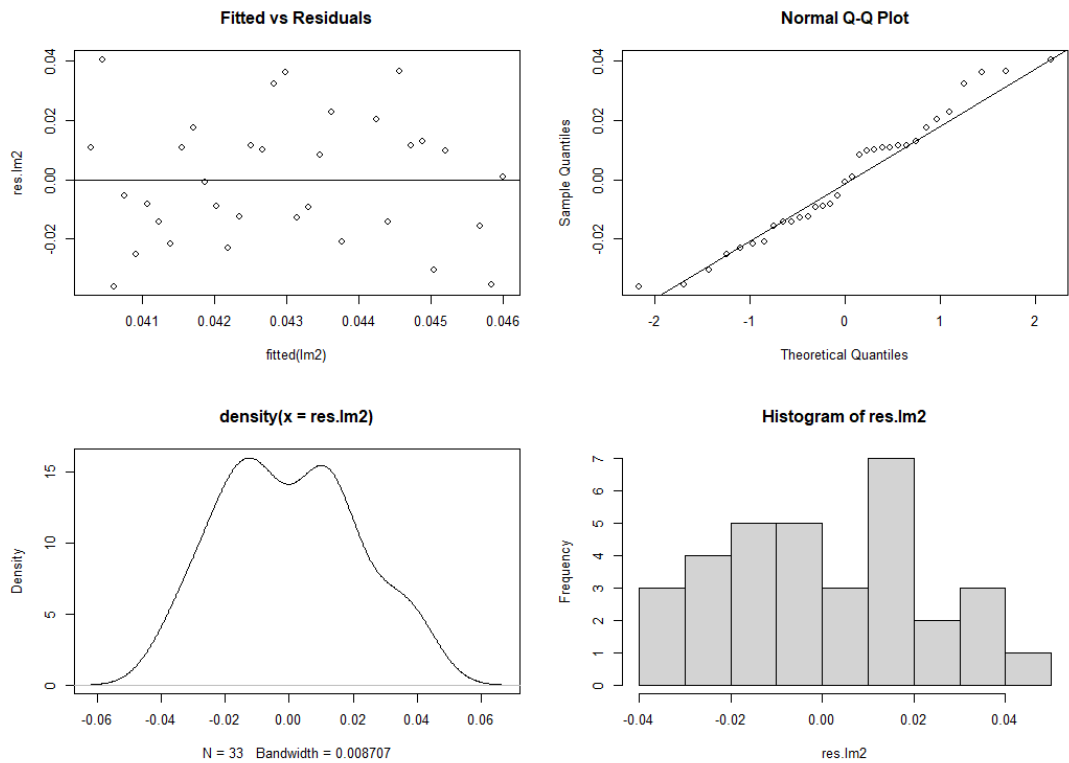


Figure A6.1. Diagnostic plots for the linear model of annual proportion of calves (up to one year old) encountered in the Hector's dolphin population at Te Pātaka-o-Rākohaitū/Banks Peninsula between 1986 and 2022.

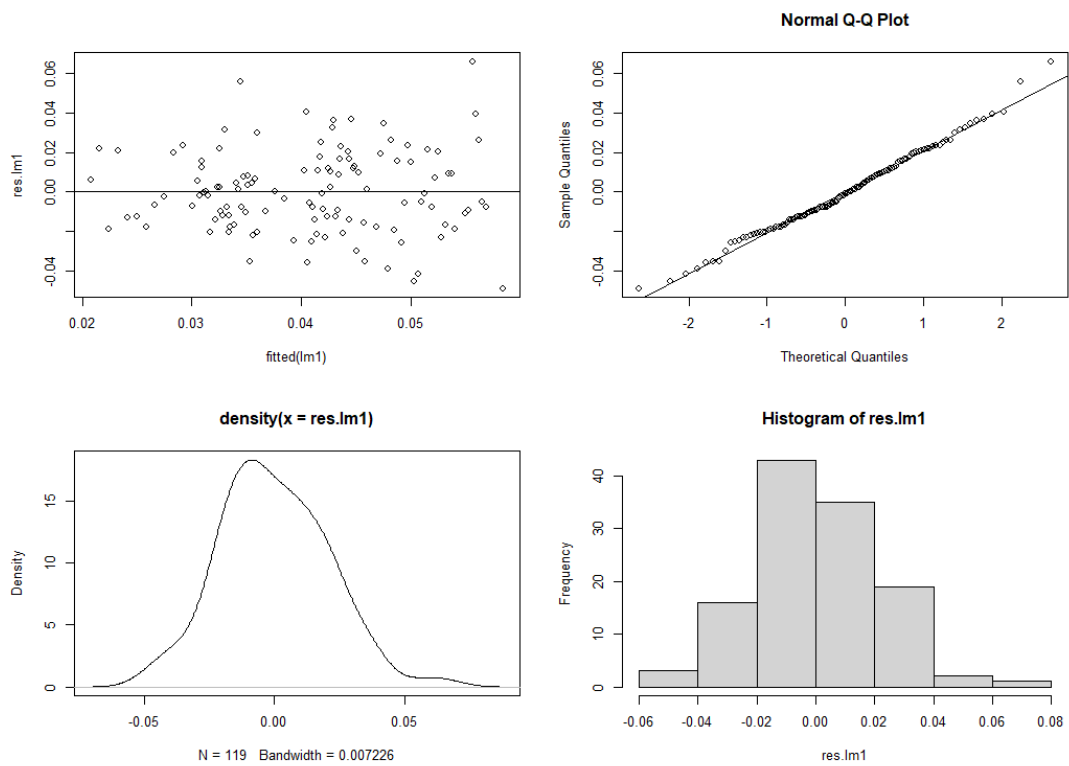


Figure A6.2. Diagnostic plots for the linear model comparing annual proportion of calves (up to one year old) encountered in the Hector's dolphin population at Te Pātaka-o-Rākohaitū/Banks Peninsula in four different survey areas between 1986 and 2022.

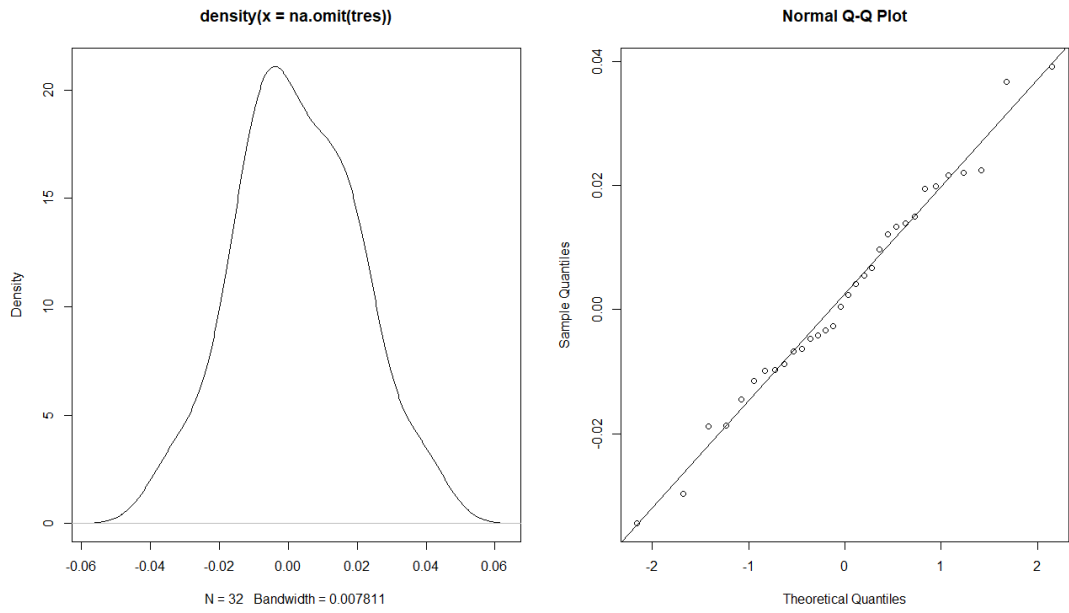


Figure A6.3. Diagnostic plots for the paired t-test comparing proportion of calves groups of Hector’s dolphins sighted in Akaroa Harbour versus the remainder of the study area at Te Pātaka-o-Rākaihautū/ Banks Peninsula between 1986 and 2022.

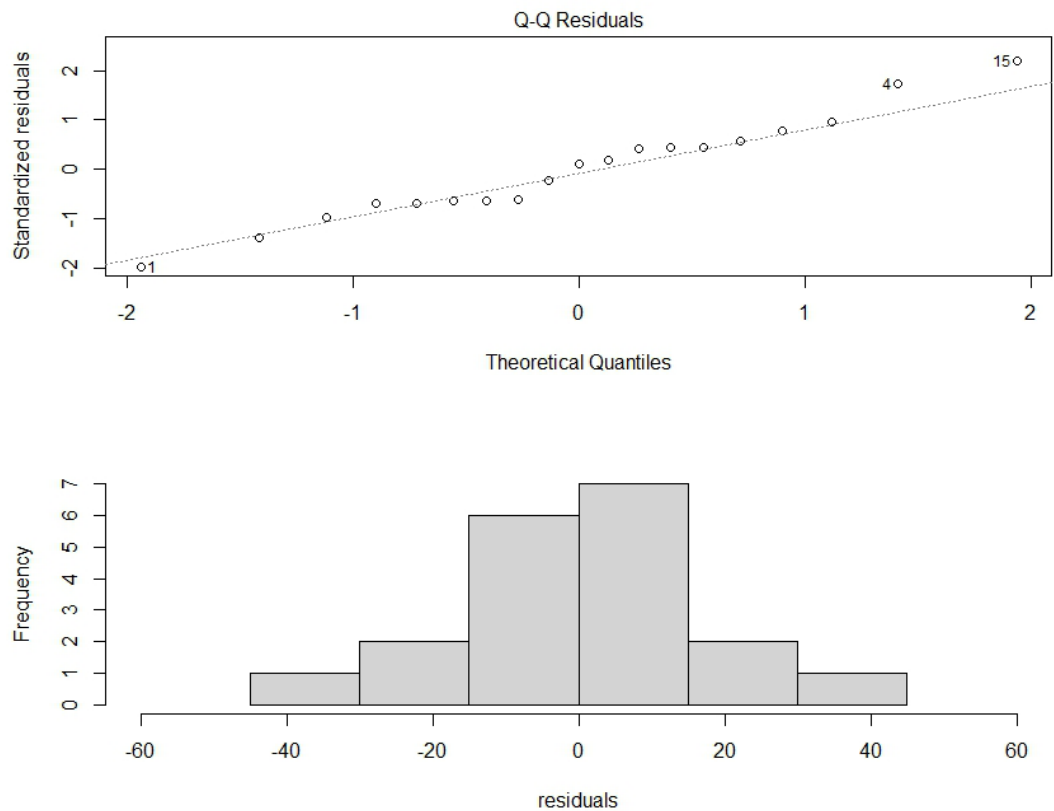


Figure A6.4. Diagnostic plots for the linear model of abundance of marked individual Hector’s dolphins over time at Te Pātaka-o-Rākaihautū/Banks Peninsula from 2002 to 2020.

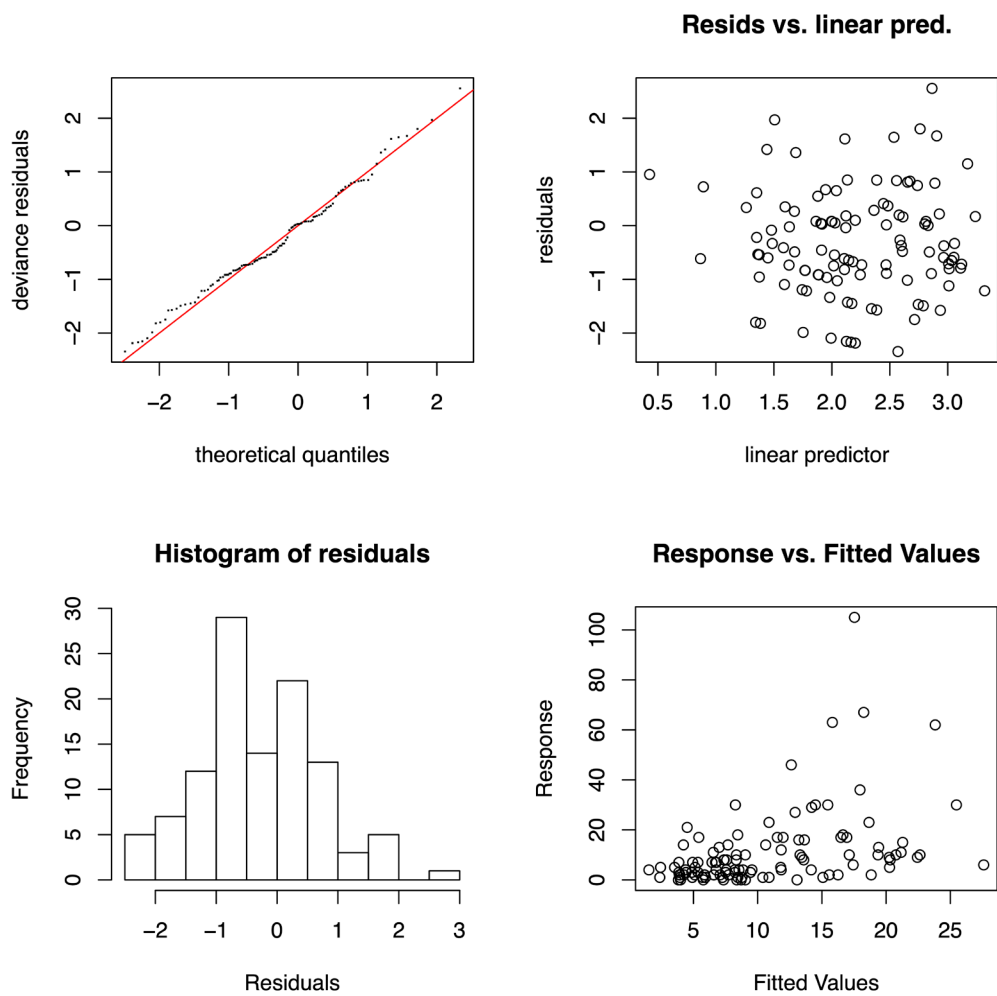


Figure A6.5. Diagnostic plots for the top generalised additive model of detection positive minutes (DPMs) per day (08:00–18:00) at Nine Fathom Point, Akaroa Harbour, from 3 December 2019 to 3 May 2020 (including during the COVID-19 lockdown) with respect to vessel traffic and swell parameters.

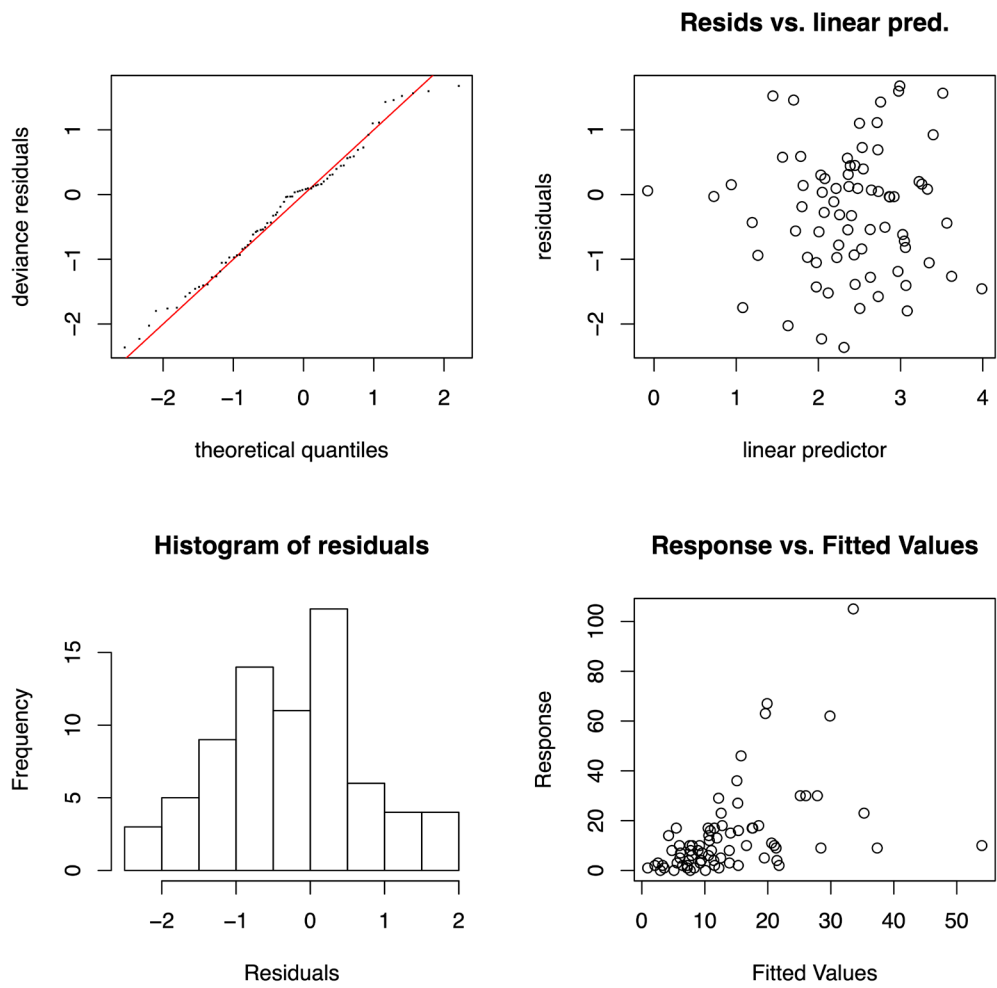


Figure A6.6. Diagnostic plots for the top generalised additive model of Detection Positive Minutes per day (08:00–18:00) at Nine Fathom Point, Akaroa Harbour, from 3 December 2019 to 23 March 2020 (excluding lockdown) with respect to vessel traffic and swell parameters.