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Hector's and Maui dolphin bycatch following 2020 management decisions

Elisabeth Slooten and Stephen Michael Dawson

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Elisabeth Slooten and Stephen Michael Dawson, Departments of Zoology and Marine Science, University of Otago, Dunedin, New Zealand.

ABSTRACT

In 2020, the New Zealand Minister for Primary Industries revised fisheries regulations with the goal of reducing Hector's and Māui dolphin bycatch. The Minister's decision was based on the Ministry for Primary Industries' (MPI) risk analysis which was presented at the 2019 Scientific Committee meeting. The peer review of the risk analysis outlined in the 2019 SC report has not yet occurred. Here we review bycatch estimates resulting from the MPI risk analysis and explore the likely level of bycatch following the 2020 management decision. As outlined below, MPI's risk analysis combines several sources of information that result in under-estimating the level of bycatch, over-estimating the effectiveness of the management decisions taken in 2020 and the species' ability to continue to sustain impacts. Some of the abundance estimates used in the risk analysis appear to be biased high, are multiplied by a reproductive rate that is much higher than published estimates and an assumed figure for calf survival. This results in over-estimating the total number of dolphin deaths each year, from which MPI subtract their estimate of bycatch which is almost certainly biased low. The remaining number of dolphins is apportioned a cause of death according to autopsy data from 55 Hector's and Māui dolphins found dead on beaches. This is then compared to estimates of what level of takes would be sustainable, calculated using a formula (PST) that is not well understood and much less conservative than the international standard (PBR, developed by the US National Marine Fisheries Service). A further problem with the MPI approach is that its definition of "risk" does not relate to the risk of population decline or extinction, and is inconsistent with the modern understanding of the behaviour of meta-populations. Instead, it defines risk as the likelihood of capture, which is apportioned to different areas according to fishing effort and predicted dolphin distribution based on a habitat model. As a result, protection has been targeted to areas where high densities of dolphins and high fishing effort coincide. Large populations were allocated the highest level of protection, while small populations remain poorly protected. This approach is likely to increase the risk of local extinctions, contractions of dolphin distribution, population fragmentation, loss of genetic variability and result in increased risk to the species as a whole.

INTRODUCTION

In 2020, New Zealand's Minister for Primary Industries revised fisheries regulations with the goal of reducing bycatch of Hector's and Māui dolphins. He chose from a range of management options presented in the Hector's and Māui Dolphin Threat Management Plan (TMP 2019) after discussion with the Minister of Conservation. Those options were based on the Ministry for Primary Industries' (MPI) risk assessment that was discussed at the 2019 Scientific Committee meeting (Roberts et al. 2019).

A key element of the MPI approach has been to fill in gaps in survey data with a habitat model. For reasons described below, this has failed to produce a scientifically robust basis for estimating the overlap between dolphins and fishing, and hence the number of dolphins caught each year and the likely effectiveness of the current regulations for managing bycatch.

In this paper, we outline the management changes in 2020 (Section A) and problems with the science underpinning the current protection measures (Section B). We also assess the likely effectiveness of the current protection measures. We find that flaws in the MPI assessment combine to under-estimate risk, resulting in a final management decision (Figure 1) that was not informed by robust science and is likely to be much less effective than assumed. The much lower level of protection for the smallest, most vulnerable populations of Māui and Hector's dolphin is a high-risk strategy.

A. THE THREAT MANAGEMENT PLAN AND CURRENT PROTECTION

The extinction of baiji (Turvey et al. 2007) and the critical situation of vaquita (Jaramillo-Legorreta et al. 2019) indicate that Critically Endangered small cetaceans need highly effective protection in order to recover to non-threatened status. For Hector's and Māui dolphins the obvious solution would be to use dolphin-safe fishing methods (i.e. other than gillnetting and trawling) throughout their habitat, out to the 100m depth contour (IUCN 2012) or 20 nautical miles offshore (IWC 2018). Māui dolphins are currently protected out to 4, 7 or 12 nautical miles offshore, depending on location. Gillnetting continues around most of the North Island and within harbours that are part of Hector's and Māui dolphin habitat (e.g. Rayment et al. 2011). Trawling continues in all but a small fraction of Hector's and Māui dolphin habitat.

For Māui dolphin, the most effective management option in the Threat Management Plan would have banned gillnets and trawling in waters less than 100 m deep in part of their range on the west coast of the North Island, with less protection off the southeastern coast of the North Island and none on the east coast. Neither the IWC recommendation (IWC 2018), nor the IUCN (2012) recommendation were included in the options presented to the public and decision makers (TMP 2019).

The TMP outlined a Toxoplasmosis Action Plan as part of the Department of Conservation's (DOC) response. While several Hector's and Māui dolphins have died of toxoplasmosis (Roe et al. 2013), the impact of this disease appears to have been substantially overstated in the TMP (see Section B below). While eliminating cats would bring many conservation benefits (e.g. for urban bird populations), it is not clear whether eliminating cats would materially benefit Māui or Hector's dolphin. Further, public opposition is likely to render this unachieveable. Focussing on wild cats, as has been suggested by DOC in public meetings, ignores the fact that abandoned domestic cats become wild cats. While a domestic population remains, there will be a ready source of recruits to the wild population. For these reasons, we see the currrent toxoplasmosis action plan as having a low chance of success. It is important to ensure that efforts to study and manage toxoplasmosis do not divert resources from addressing direct impacts that can be readily managed, such as fisheries bycatch.

The most effective option in the TMP involved extending existing protection from gillnet bycatch in a few areas, while leaving important regional populations off southeast, northeast and west coasts of the South Island with little or no protection. These populations have had recent bycatch mortalities, and low abundance in several of these areas renders them especially vulnerable to decline. The Ministers were not provided with any management options for improving protection off the South Island west coast or North Island east coast.

Other areas left unprotected include a gap off the south-east coastline of Banks Peninsula, a well documented hotspot of Hector's dolphin abundance (e.g. Rayment et al. 2010; MacKenzie and Clement 2014; Brough et al. 2020). This omission results directly from the habitat model (see section B below) and the assumption that fishing effort will not be displaced in response to new protection measures. Past experience shows that fishing effort will be displaced to areas where fishing restrictions are less stringent. Gaps in protection are likely to become the focus of intense fishing activity. This is likely to increase bycatch off the south-eastern coast of Banks Peninsula, resulting in the area becoming a population sink. This would shift, rather than solve, the bycatch problem.

The distribution of fishing effort can change dramatically over time, particularly in response fisheries regulations. For this reason, efforts to solve bycatch problems based on the distribution of current or recent fishing effort are likely to have poor long-term effectiveness. When fishers shift their area of operation, bycatch problems shift with them. This problem has been recognised for decades (e.g. Dawson 1991). The solution is to ensure there is effective protection where the dolphins are, rather than only where gillnet and trawl fisheries are currently operating. Closing an area to gillnet or trawl fisheries has no economic cost if gillnet and trawl fisheries are not operating in the area currently, and protects dolphin populations from shifts in fishing effort, whether caused by regulation, stock depletion or market pressure.

Possibly the most serious oversight is the lack of attention given to small, local and regional populations (Taylor et al. 2018). Photo-ID and genetic studies have shown small populations, off the Catlins, Otago and in Cloudy Bay, to be resident, with reasonably precise mark-recapture estimates of abundance (Turek et al. 2013, Hamner et al. 2017, Harvey, unpub. data). These local populations connect the larger populations in Te Waewae Bay and Canterbury, and provide a conduit for genetic and demographic connectivity. If they decline to extirpation, the current fragmentation of the species' total population (Hamner et al. 2012) will increase, increasing risk to the species as a whole.

For the reasons outlined above, the protection measures in the TMP, and especially the final decision by the Minister for Primary Industries, are inadequate. Even the best options in the TMP (2019) would have been unlikely to be effective in reducing bycatch to sustainable levels – let alone allow population recovery. Unfortunately, the protection afforded by the final Ministerial decision fell well short of the best options in the TMP.

B. PROBLEMS WITH THE SCIENCE SUPPORTING THE TMP

The Ministry for Primary Industries' (MPI) threat assessment approach (Roberts et al. 2019) relies critically on information on dolphin distribution, population size, reproductive rate, fishing effort and the catch rate of dolphins per unit of fishing effort. This section briefly outlines the main problems with the approach taken, and the data inputs used in the risk assessment (Roberts et al. 2019) that led to the TMP and the Ministerial decision.

1. Distribution and abundance

The population survey carried out under contract to MPI (MacKenzie and Clement 2014, 2016, 2019) resulted in robust information on dolphin distribution in high density areas. For example, it confirmed that Hector's dolphins range throughout waters less than 100 metres deep with a few sightings in deeper water. Unfortunately, the survey intensity was insufficient to provide data on dolphin distribution in low-density areas, including small populations and offshore areas. This is problematic because the smallest populations are at greatest risk. For example, the MPI survey made no sightings off Otago where there is a population of about 42 Hector's dolphins (CV 41%, CI 19 – 92; Turek et al. 2013). MacKenzie and Clement (2014) made only one sighting off the north coast of the South Island, where the population is estimated at some 200 Hector's dolphins. It is not possible to determine the distribution of 200 dolphins on the basis of one sighting.

These and other problems with population surveys carried out under contract for MPI have been pointed out in several peer reviews (including IWC 2016). Comparison of the east coast South Island estimate with previous line-transect surveys (Dawson et al 2004) and local genetic and photo-ID mark recapture estimates (Hamner et al 2017; endorsed by the Scientific Committee, IWC 2017), suggest that, despite methodological differences, the MPI estimate is biased high.

In most areas, the MPI surveys (MacKenzie and Clement 2014, 2016, 2019) produced similar population estimates to previous surveys (Dawson et al. 2004; Slooten et al. 2004, 2006). Off the west coast of the South Island, the most recent population estimate is about 1000 individuals fewer than the previous estimate (Slooten et al. 2004). A decline of this order was predicted by Population Viability Analysis (e.g. Slooten and Dawson 2010) due to the poor protection on the west coast. Population surveys of Māui dolphin by Otago and Auckland Universities likewise show a declining population, from 111 in 2004 (Slooten et al. 2006) to 57 currently (Cooke et al. 2019). Wade et al. (2012) and Cooke et al. (2019) estimated declines on the order of 3% per year, based on boat surveys, aerial surveys and genetic Mark-Recapture data.

The population surveys by MacKenzie and Clement (2014, 2016, 2019) were optimised for estimating total population size, rather than dolphin distribution. Therefore, survey effort was low in areas where few dolphins were expected. To fill in these gaps, Roberts et al. (2019) developed a habitat model to predict dolphin distribution.

2. Habitat model

Roberts et al. (2019) had access to very limited in-situ environmental data. Their habitat model uses the presence or absence of particular fish species (collected by bottom trawl), and satellite data on water turbidity by season. Māui and Hector's dolphins are often found in turbid water (e.g. Bräger et al. 2003). While turbidity could potentially act as a proxy for another environmental variable (e.g. prey availability), there is no evidence to suggest that turbidity *per se* has a direct effect. The hypothesis that turbidity is important to Hector's dolphins does not explain how the species apparently managed to flourish before large scale deforestation and intensive land use created the current levels of turbidity.

In the modelling process the dolphin abundance data (MacKenzie and Clement 2014, 2016, 2019) had a seasonal signal. Hector's dolphin distribution is more dispersed in winter than in summer, with respect to water depth and distance from land (Rayment et al. 2010; MacKenzie and Clement 2014, 2016, 2019). Turbidity was modelled with a seasonal component, while fish distributions were modelled as year-round averages. It was therefore not surprising that the habitat model suggested a correlation between dolphin distribution and turbidity. This is not an indication that turbidity drives dolphin distribution, but an indication that the modelling process was flawed. A better fit is likely to result from including depth and season (summer, autumn, winter, spring) directly in the model or using seasonal fish data.

Depth is *directly* relevant to Hector's dolphins, because they feed mostly on bottom-dwelling fish (Miller et al. 2013).

There are also problems with the manner in which the statistical modelling was done:

- It did not meet usual standards for model checking and diagnostics (e.g. Redfern et al. 2006, Elith and Leathwick 2007, Guillera-Arroita et al. 2015). For example, the standard practice, of developing the model with part of the dataset and testing it with the remaining data, was not followed.
- The fish data used were presence/absence only, with no accounting for differences in sampling method. Mid-water and epipelagic species are important in Hector's dolphin diet (Miller et al. 2013), but are not sampled effectively by the bottom trawl method used by NIWA to collect the data used in the habitat model.
- Model choice was not based on a quantitative and/or objective measure of model fit (e.g. Akaike's Information Criterion), but instead based on feedback from a stakeholder workshop. Evidence ratios (Anderson 2008), calculated directly from the AIC scores, show that the model chosen was 10.6 trillion times less likely than the best-fitting model.

The fit between the habitat model and the sightings data (from research surveys and public sightings; Roberts et al. 2019) has not been estimated statistically and appears to be poor (Figure 2). As a result, current protection for Māui dolphin does not include the east coast of the North Island, despite consistent sightings there.

Hector's dolphin distributions predicted by the habitat model do not match rigorously collected survey data (e.g. Rayment et al. 2010, MacKenzie and Clement 2014). This is problematic for an approach that estimates bycatch based on the overlap between dolphins and fishing.

3. R_{max}

Estimates of how many dolphin deaths in fishing nets can be sustained depend strongly on R_{max}. Under contract to MPI, Edwards et al. (2018) adjusted the existing estimate of R_{max} upwards, using a curve of the relationship between body size and reproductive rate across a wide variety of mammals (Duncan et al 2007). There are five problems with this:

- The relationship illustrated in Duncan et al. (2007) reflects a very general emerging pattern that is not expected to hold under all circumstances. Exceptions are entirely valid.
- Hector's dolphin would be *expected* to be an outlier. It is one of world's smallest dolphins, living in cool temperate waters in which calves must be proportionately large and well insulated to survive. Therefore, breeding females must invest a large amount of energy into each calf, which is likely to reduce the rate at which they can produce calves.
- When a data point does not fit an expected relationship, it is not legitimate to move the data point. The normal science process involves fitting lines to data points, not data points to lines.
- There is no empirical evidence to support the revised-upwards estimate of R_{max} for Hector's dolphins. Indeed the 2019-2020 field season produced the lowest ratio of calves to non-calves (0.8%) recorded since 1991 (Dawson and Slooten unpub. data).
- The use of a general relationship between body size and R_{max} is an appropriate last resort only for a species for which no relevant biological data are available. It is not an argument for ignoring existing biological data.

4. Fishing effort data

The available data on the location of fishing effort are very limited, and unlikely to be representative of the fishery as a whole. Despite GPS being widely available since the early 1990s, fishermen have only recently been required to provide GPS data (Figure 3). The quality of these location data has been questioned. A fishing industry representative taking part in the Expert Panel workshop in 2018 (Taylor et al. 2018) pointed out that the current effort data are biased by larger vessels being more likely to report the GPS locations of their fishing effort. These data are apparently taken at face value in MPI's estimation of bycatch.

It is not clear how Roberts et al. (2019) extrapolated the spatial distribution of current or past fishing effort from the limited spatial information available. Until 2005, GPS locations were reported for less than 1% of the gillnet fishing effort. This increased to just over 15% in 2006, 56% in 2007, and around 60 - 65% since (Figure 3). The remaining fishing effort is reported by large fisheries statistical areas (Figure 4).

In the absence of accurate data on the location of fishing effort, Davies et al. (2008) attempted to estimate the overlap between dolphins and fishing by selecting fishing effort based on target fish species. Davies et al. (2008) excluded gillnet effort targeting yellow-eyed mullet, grey mullet and flounder, arguing that these nets were set in waters too shallow to catch Hector's dolphins. They also excluded gillnet effort targeting ling, bluenose, groper, tarakihi and warehou, arguing that these are set in waters too deep to result in Hector's dolphin bycatch. These assumptions are now known to be incorrect. The most recent population surveys (Rayment et al. 2010, MacKenzie and Clement 2014) show greater offshore extent in dolphin distribution than was assumed by previous researchers, and therefore potentially greater overlap between Hector's dolphins and gillnetting. Davies et al. (2008) reduced the total amount of fishing effort used in their calculations to about half of the fishing effort actually overlapping with Hector's dolphins.

Roberts et al.'s (2019) risk analysis also under-estimates the overlap between dolphins and fisheries because their habitat model dramatically reduces the apparent offshore distribution of the dolphins and fails to take account of dolphin movement between protected and unprotected areas. The poor fit between the habitat model and the actual sightings from MacKenzie and Clement's surveys (2014, 2016, 2018) and previous surveys (Rayment et al. 2010) indicates that bycatch estimates would be much higher if they were based directly on sightings data from population surveys.

Despite the negative biases outlined above, the most recent bycatch estimate for the Banks Peninsula to Timaru area (37 dolphins per year; TMP 2019) is more than twice the estimate from the 1997-1998 observer programme (16 dolphins per year; Baird & Bradford 2000). This suggests that past bycatch estimates (Baird & Bradford 2000; Davies et al. 2008) were substantial under-estimates of actual Hector's dolphin bycatch. Davies et al. (2008) estimated bycatch during 2000-2006 at 110-150 Hector's dolphins per year. From 1988-2008 there was only one, relatively small, protected area (Figure 1). A substantial increase in protection in 2008 should have reduced 2008-2020 bycatch to less than half of the number of dolphin deaths per year compared to 1988-2008 (Slooten and Davies 2011). Instead of halving, the bycatch estimate for the only area with any appreciable observer coverage is estimated to have more than doubled. If 37 dolphins were caught each year during 2008-2020, this suggests that the 1988-2008 bycatch could have been >60 per year in the Banks Peninsula-Timaru area and 200-300 per year around NZ based on the change in overlap between dolphins and fishing alone. In addition, the total amount of fishing effort has changed over time.

The TMP estimate of current bycatch is 59 Hector's dolphins per year (95% CI 23-127). Current gillnet fishing effort in Hector's and Māui dolphin habitat is about 8,000 fishing days per year (Figure 5, compared to about 20,000 days per year during the 1990s and 35,000 days per year in the early 1980s. If there had been no change in the size of the dolphin population, and no change in the spatial distribution of fishing effort, one would expect about 150 dolphin deaths per year during the 1990s and early 2000s, and about 260 dolphin deaths per year during the early 1980s, based on the higher fishing effort alone. Bycatch of 260 Hector's dolphins per year is on the order of 3 times the PST, and 23 times the PBR for the current dolphin population. Population size in the 1980s, and therefore bycatch, is likely to have been substantially higher than today, as estimated in previous risk assessments (e.g. Slooten and Dawson 2010, Davies et al. 2008).

In addition to much higher fishing effort in the 1980s and 1990s, the overlap between dolphins and gillnet fisheries was also much greater. Gillnetters fished closer to shore (Dawson 1991) and there was no protection before 1988. Before 2008, the only areas with dolphin protection measures were Banks Peninsula and the North Island west coast. Since 2008, gillnets have been banned from the shoreline to 4 nautical miles offshore off most of the east and south coasts of the South Island.

5. Observer coverage and estimates of bycatch

Observer coverage in New Zealand's inshore fisheries is exceptionally low, and patchy in space and time (Figures 5 and 6). The only robust observer progamme for quantifying bycatch of Hector's dolphins was achieved during the 1997-1998 fishing season (Baird and Bradford 2000). Since then, annual observer coverage has been less than 5% in the gillnet (Figure 5), and inshore trawl fleet (Figure 6). The number of observer days per year in gillnet fisheries overlapping with Hector's and Māui dolphins increased briefly after the last TMP (2007) and then dropped again to fewer than 5 observer days per year during 2011-2018 (Figure 5). MPI estimate that trawl fisheries currently catch around 14 Hector's and Māui dolphins per year. Yet there has only been one observed capture to date (in the 1997-1998 fishing season).

The real problem is not that the bycatch estimates are highly uncertain, but that sampling theory suggests that they are likely to be biased low. For any one fishing vessel, catching a Hector's dolphin is a relatively rare event. Low observer coverage acts to under-estimate bycatch.

We investigated this bias quantitatively via bootstrapping in R, for a gillnet fishery with 1000 gillnet sets in a season, 1000 years of monitoring and an underlying catch rate of 15 dolphins per 1000 gillnet sets (Figure 7). The fishing effort in the bootstrapping exercise is about double the size of the Timaru gillnet fishery in the 1997/98 observer programme. The catch rate used is lower than that found in the 1997/98 observer programme, but higher than that assumed by MPI to occur currently.

Notwithstanding significant levels of actual catch, at low (but typical) levels of observer coverage, the likelihood of observing a catch is also very low. With the 2% observer coverage in the bootstrapping exercise above, there is a 75% chance of seeing no captures in any one year (Figure 8).

Extending these trials to cover a wider range of values for observer coverage and plausible bycatch rates shows that the very low level of recent observer coverage (1-3%) has resulted in estimates of bycatch rate that are almost certainly biased low (Table 1).

The US guidelines for preparing stock assessment reports (GAMMS 2016) address this problem specifically. In their terminology, the observer coverage applied in NZ inshore fisheries results in bycatch estimates that are "always biased". This is a fundamental sampling problem that is not solved by the Bayesian approach taken in the analysis by Roberts et al. (2019).

That bycatch has been underestimated is consistent with Cooke et al.'s (2019) analysis of Māui dolphin population trends, in which the best fitting models suggested levels of bycatch 15-20 times higher than those assumed by Roberts et al. (2019).

All observed catches have occurred at Kaikoura (area 18), north of Banks Peninsula (area 20) or south of Banks Peninsula down to Timaru (area 22). All of these areas have relatively high fishing effort. Average reported fishing effort during 2015-2018 was 390 km of gillnet / year on the north side of Banks Peninsula (area 20) and 982 km / year on the south side of the peninsula to Timaru in the south (area 22). Most of the observed captures (13) have been in the Banks Peninsula area, which has relatively high fishing effort and high dolphin densities. A smaller number of captures (4) have been observed in Kaikoura (area 18), which has the highest fishing effort recorded in any of the fisheries statistical areas, with 1,372 km of gillnetting effort per year during 2015-2018, but a relatively small population of a few hundred Hector's dolphins.

In several areas, the timing of observer coverage has reduced the probability of detecting bycatch, with low or no observer coverage during years of high fishing effort. For example, area 34 (on the west coast of the South Island, Figure 4) saw high fishing effort (c. 1200 km / year) throughout the late 1980s and early 1990s. The only observer coverage in this area, however, has been 11 observer days in 2009 and 10 observer days in 2013, when fishing effort was below 100 km of gillnet per year.

A substantial number of multiple captures, of several Hector's dolphins caught in the same gillnet or trawl (Slooten et al. 2019), makes it even more difficult to obtain robust estimates of bycatch. Of the 8 dolphins caught in the first observer programme (Baird and Bradford 2000), 4 were single captures and 4 were caught in multiple captures (2 capture events, each catching 2 dolphins in one gillnet). Recent voluntary reports also include a substantial proportion of multiple captures, including 5 Hector's dolphins caught in one gillnet on 19 February 2018, 3 Hector's dolphins caught in one trawl net on 20 December 2018 and 3 Hector's dolphins caught in one trawl net on 18 February 2019 (Slooten et al. 2019, DOC 2021b). Voluntarily reported captures are not used in the MPI estimations of bycatch. Both the number of multiple captures and their locations are inconsistent with the latest government estimates (Roberts et al. 2019).

MPI estimate that there are currently 59 Hector's dolphins killed each year in commercial gillnet and trawl fisheries (95% ci 23-127; TMP 2019). Yet observer coverage is so low that only 8 Hector's dolphin captures have been observed in the last 20 years, each of a single dolphin caught in a gillnet. A cryptic mortality multiplier is used to correct for unobserved mortality (e.g. drop-out). However, with observer coverage so low that zero bycatch is observed in most areas and most years, this multiplier is rarely applied. No robust estimates are available for dolphin deaths in recreational gillnets, 'customary' gillnets or ghost nets or illegal fishing.

Since November 2019, on-board cameras are required on gillnet and trawl vessels between 8 and 29 m long, fishing in Māui dolphin habitat to 12 nautical miles offshore. The lower size limit appears to be driven by the difficulty of fitting such systems to the small boats (<8 m) commonly used to set gillnets in the harbours in Māui dolphin habitat. Video camera monitoring is required only in the orange areas in Figures 9 and 10. In this area, 20 vessels have opted in, with the remaining eight vessels promising not to fish in the areas to be observed. The 20 video monitored vessels represent a small fraction of the total gillnet and trawl vessels operating in Māui dolphin habitat. Currey et al. (2012) reported 120-209 gillnet vessels and 35-67 trawlers operating in this area. The TMP (2019) estimated that up to 160 gillnetting vessels and 18 trawlers would have been impacted if the Minister of Fisheries had chosen the most effective option in the TMP. The many vessels fishing in the harbours are not monitored. That Māui dolphins use these harbours is evident from sightings, in-situ passive acoustic monitoring and bycatch (e.g. Rayment et al. 2011, DOC 2021b). In 2002, for example, a fisherman caught two Māui dolphins in one gillnet inside the Manukau Harbour (DOC 2021b).

Observer coverage of 95.5% for gillnet fisheries and 88.3% for trawl fisheries, reported to the Scientific Committee (IWC 2018), applies only to the orange areas in Figures 9 and 10. Observer coverage is below 5% in most areas and years (Slooten et al. 2019, Slooten and Dawson 2017) due to the many exceptions based on area, vessel size, etc. While video monitoring has promise, a robust commitment from management agencies to actually review the footage is needed for this strategy to be credible. For example, two Hector's dolphins were observed from 31 gillnet hauls in a camera trial during 2012-2013. The footage, however, was viewed only after the skipper reported a dolphin capture. Of the remaining 5 fishing vessels participating in the video monitoring programme, on average <10% of the footage was viewed (MPI 2013).

Neither cameras nor human observers detect 100% of dolphins caught, and a combination of observers and on-board cameras would be required to estimate the number of dolphin deaths missed due to drop out and other factors (e.g. observer not on deck when net is hauled). This is practical only in areas with high fishing effort and high dolphin densities. Even with 100% video or observer coverage, bycatch estimation will remain problematic for very small populations (e.g. North Island and north and south coasts of the South Island). In these areas there is a high risk of failing to detect bycatch, and being unable to determine if the PBR / PST has been exceeded.

6. PST - the metric used to determine a sustainable take.

The Potential Biological Removal (PBR) method has been thoroughly tested via simulation (Wade 1998). A method developed in New Zealand, in the context of seabird bycatch, known as the Population Sustainability Threshold (PST) is being used to assess Hector's and Māui dolphin bycatch. The PST formula uses ϕ, a so-called tuning parameter, that conflates uncertainty and the recovery objective. The PST approach has had limited peer review, and has not been published in a peer-reviewed scientific journal. It has not been subjected to the extensive testing via simulation that the PBR has, and it is not clear if the resulting bycatch limits are sustainable. In most cases PST estimates are substantially larger than PBR estimates (Table 2).

The US approach was developed by a team of marine mammal scientists who are well aware of meta-population theory, population fragmentation and source-sink dynamics. It therefore includes strategies to avoid depletion of local populations (Wade 1998). For example, the PBR approach requires limits to be calculated for populations that are demographically separate from others (i.e. a dolphin removed from one population will not likely be replaced from another population). The New Zealand approach includes none of these safeguards, with PST calculations routinely made for very large areas with several dolphin populations and several fisheries. The relative performance of the PBR and PST needs to be thoroughly tested to determine whether the PST results in genuinely sustainable bycatch limits.

In addition, the New Zealand bycatch monitoring system would need to be substantially improved to make it possible to set and monitor the relatively small bycatch limits for individual populations (Table 2). This is most clearly seen in the management of Māui dolphin bycatch. Both PST and PBR are on the order of 0.1 (one dolphin every decade) for Māui dolphin. Even with 100% observer coverage, some dolphin deaths will not be observed (e.g. because the dolphin drops out of the net before being detected by an observer or video camera).

It is clear that the range of Hector's and Māui dolphin has been substantially reduced by past human impacts (Dawson et al. 2001, McGrath 2020). For example, beachcast carcases and dolphin sightings off the North Island east coast (Figure 2) include 59 sightings, with 38 since 2008 and several in 2019 and 2020, indicating that this area is part of the species' current range (e.g. Cawthorn 1988, DOC 2019, 2021a, 2021b, Russell 1999, Pichler 2002, McGrath 2020). It is also an area of substantial fishing effort (Figures 9 and 10), extremely low observer coverage and no protection. The very small remaining dolphin population in this area is at high risk of extirpation. In this context it is crucial to consider whether population targets should be defined in terms of estimated carrying capacity for the whole of the North Island or just for the few areas where Māui dolphins are still relatively abundant today. To be biologically meaningful, population recovery requires recovery and recolonisation of the original range. It is unclear how both aspects of recovery would be provided for in PST calculations. Another critical problem is that in the TMP, the PST is used as a limit on fisheries mortality alone, when it (and the PBR) were designed as a limit on total human impact (Richard et al 2017).

7. Movement and the potential for Source-Sink dynamics

Failing to take into account inshore/offsore and alongshore movement of dolphins between protected and unprotected areas leads to further under-estimation of bycatch and over-estimation of the effectiveness of current protection measures. This results from a mis-match between the area for which bycatch is estimated using the habitat model and the area for which a bycatch limit is set using the PST (total number of dolphins in protected inshore areas and unprotected offshore areas). This will cause dolphin numbers to decline, initially offshore as the bycatch limit is too high for the number of dolphins in the unprotected area offshore. Over time, the number of dolphins in the protected area also declines, because dolphins continue to use the whole area – including inshore and offshore areas. The exact outcome will depend on the catch rate and the rate of dolphin movement between protected and unprotected areas. However, it is clear that dolphin movement will increase bycatch compared to assuming that dolphins in protected areas are completely protected.

A second form of source-sink dynamics involves dolphin movements among neighbouring populations. For Hector's dolphins in Canterbury, the rate of movement has been estimated at below 1% p.a. (Fletcher et al. 2002). A sensitivity analysis, exploring the effect of movement rates between neighbouring populations from 0-5% p.a., show that source-sink dynamics are an important determinant of the rate of growth or decline in individual populations (Slooten 2007). Without movement, a heavily impacted, declining population does not affect nearby populations. With 5% movement, a declining population can substantially deplete neighbouring populations. Both types of source-sink dynamics appear to be missing from Roberts et al. (2019).

8. Estimating the incidence of non-fishery causes of death

Roberts et al. (2019) estimate the number of dolphin deaths from toxoplasmosis by applying the percentage of toxoplasmosis deaths in a sample of beachcast dolphins (Roe et al. 2013) to the number of dolphin deaths 'left over' after removing the estimated number of deaths due to bycatch. For example, if there are estimated to be 1000 dolphin deaths per year of which 60 die in fishing nets, that leaves 940 dolphin deaths due to other causes. The percentage of dolphins with toxoplasmosis (about a third of a small sample of dead dolphins found on beaches) is then used to estimate how many of the 940 dolphin deaths were caused by toxoplasmosis. This approach results in an estimate of 334 dolphin deaths from toxoplasmosis each year (TMP 2019).

Necropsies of dolphins that washed up on beaches were used to estimate the number of Māui and Hector's dolphin deaths from disease. This is a biased sample because most of the animals being dissected would be expected to be ill or old. In general, animals near the end of their lifespan are more susceptible to disease. Basing conclusions on a beachcast sample exaggerates the importance of disease, and cannot properly account for other contributing factors.

The argument that more than 300 Māui and Hector's dolphins die from toxoplasmosis each year is not scientifically defensible. Presenting this estimate of toxoplasmosis deaths in the TMP has led to confusion in public discussions, implying that disease is worse than fisheries mortality and therefore that bycatch does not need to be managed. The 2018 Expert Panel (Taylor et al. 2018) pointed out that there is no reason to believe that beachcast carcasses, in particular such a small sample, are representative of deaths of all kinds throughout the dolphin population. They stated "we are concerned that the results from the model could be seriously misleading. For this reason, we recommend that you [MPI] 'back off' from forcing the model to produce conclusions which are supportable only when a series of questionable assumptions are made and which even then, are highly uncertain." (Taylor et al. 2018).

The expert panel (Taylor et al. 2018) concluded that *"we are not convinced that it is appropriate for the toxoplasmosis necropsy data to receive the full modelling treatment: the uncertainties and potential biases in these data are too large. If the effects of the disease are as large as they appear, and the deaths are additional to other causes, we would expect the populations of Hector's dolphins to be in rapid free-fall towards extinction" (point 32, p.12)*

MPI's estimates of the importance of disease are also not consistent with published field observations:

- The survival rate of dolphins at Banks Peninsula increased significantly (Gormley et al. 2012) after gillnetting was banned inshore in 1988. If disease was an important problem this management change would have made negligible difference.
- The trend in genetic mark-recapture estimates of Māui dolphin also suggest a slower current rate of population decline than in the past (Baker et al. 2016). If toxoplasmosis

was as severe a problem as the TMP claims, this population should be continuing to decline rapidly.

The discussion of the impact of disease in the TMP also ignores the fact that it is, effectively, unmanageable. By contrast, bycatch is readily manageable. The two best known examples of toxoplasmosis in marine mammals are Hawaiian monk seals and sea otters. Barbieri et al. (2016) described nine Hawaiian monk seal deaths from toxoplasmosis between 2001 and 2015. They noted that during that same period, the population more than tripled in size (from about 50 to 183 individuals). Verma et al. (2018) examined 70 northern sea otters and found that 65 of them had toxoplasmosis antibodies. Nonetheless, that population grew at 9% annually during 1989-2018 (U.S. Fish and Wildlife Service 2018) and the southern sea otter has now grown to about 3,000 individuals — close to the threshold for delisting the population (U.S. Fish and Wildlife Service 2003). These examples show that the presence of toxoplasmosis does not mean it is crucially important at a population level.

The TMP (2019) suggests that toxoplasmosis kills 1.9 Māui dolphins (95% credible interval 1.1 – 3.0) and 334 Hector's dolphins per year (95% credible interval 132 – 625) and is the main cause of the decline and lack of recovery of Hector's and Māui dolphin populations. If toxoplasmosis is as serious as claimed, then total human impact on Māui dolphin is about 20 times higher than the PST. As there is no solution to toxoplasmosis on the horizon, bycatch will need to be reduced to levels approaching zero, to give Māui dolphin any realistic chance of recovering from its current Critically Endangered population level.

CONCLUSIONS:

A. The first overarching problem is that the Roberts et al. (2019) approach combines several estimates that are biased, and in combination these act to underestimate the level of bycatch and overestimate the species' ability to absorb impacts.

   In essence, the approach uses abundance estimates that are likely biased high, multiplies them by a reproductive rate that has been arbitrarily raised, multiplied by an assumed figure for calf survival, to reach a number of dolphins that would be added each year if the population were to remain stable. From this number, Roberts et al. (2019) subtract their estimates of bycatch, which are likely to be biased low. The remaining number of dolphins is apportioned a cause of death according to autopsy data from 55 Hector's and Māui dolphins found dead on beaches. This is then compared to estimates of what level of takes would be sustainable, calculated using a formula that is not well understood and less conservative than the PBR.

   This is a poor basis for rational management of an endangered, endemic marine mammal.

B. The second overarching problem is that the Roberts et al. (2019) approach to "risk" does not relate to the risk of population decline or extinction, and is inconsistent with the modern understanding of the behaviour of meta-populations. The approach defines risk as the likelihood of capture, which is apportioned to different areas according to fishing effort and the habitat model's outputs for dolphin distribution. The protection options are therefore targeted where high densities of dolphins and high fishing effort coincide. Large populations are allocated the highest level of protection, while small populations remain poorly protected.

This approach makes no sense from a meta-population point of view. When fishing effort shifts from protected areas to neighbouring, unprotected or poorly protected populations, small, local populations are likely to decline. Population fragmentation is likely to increase. This approach would increase the risk of local extinctions, contractions of dolphin distribution, loss of genetic variability and result in increased risk to the species as a whole.

Many of these points were made in the Expert Panel workshop in 2018 (Taylor et al. 2018). To our knowledge, MPI have not yet provided a list of recommendations from the Expert Panel Report with their responses (if any) to those recommendations. This standard step in scientific practice, following peer review (e.g. response to reviewers' comments) has not been followed.

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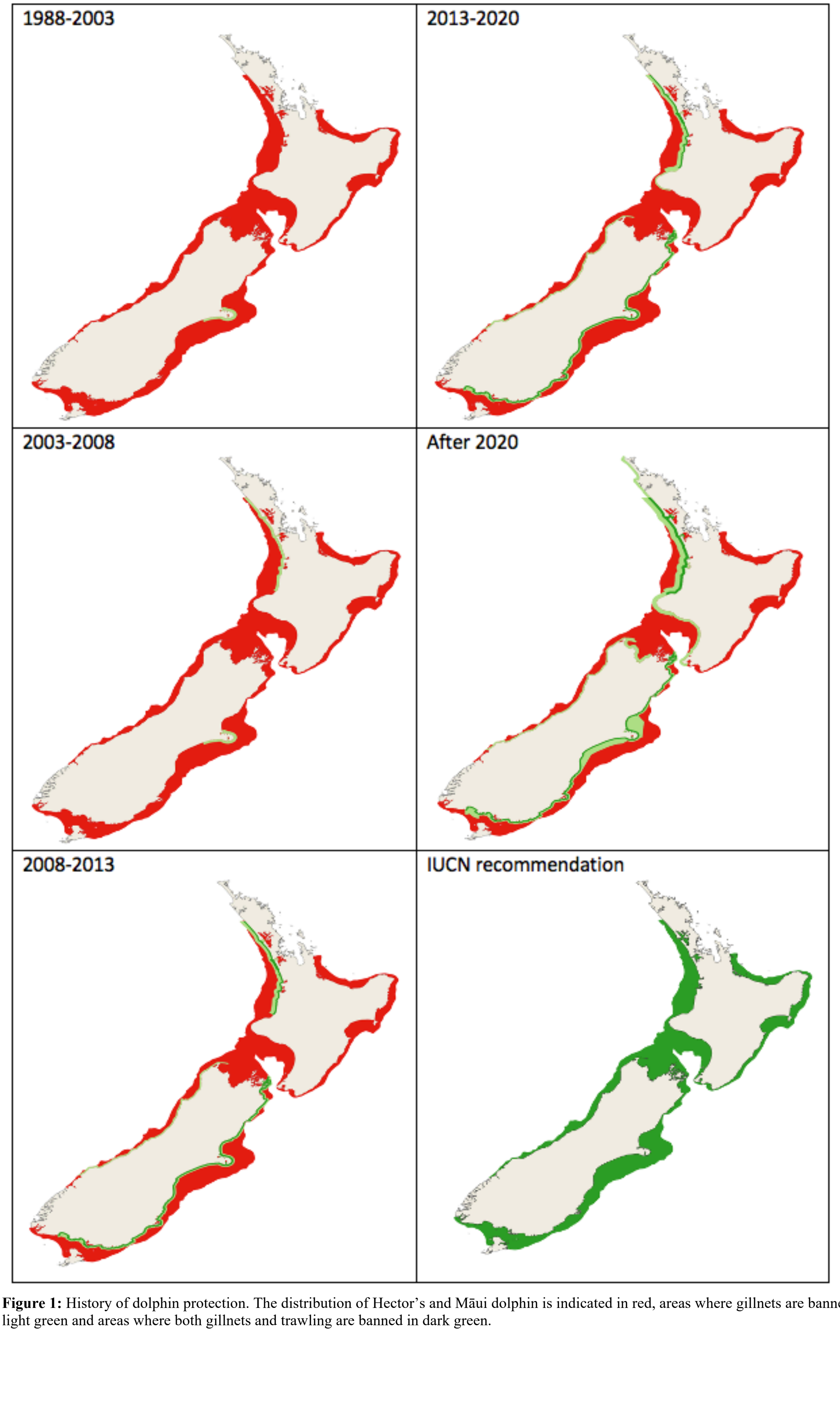

Figure 1: History of dolphin protection. The distribution of Hector's and Māui dolphin is indicated in red, areas where gillnets are banned in light green and areas where both gillnets and trawling are banned in dark green.

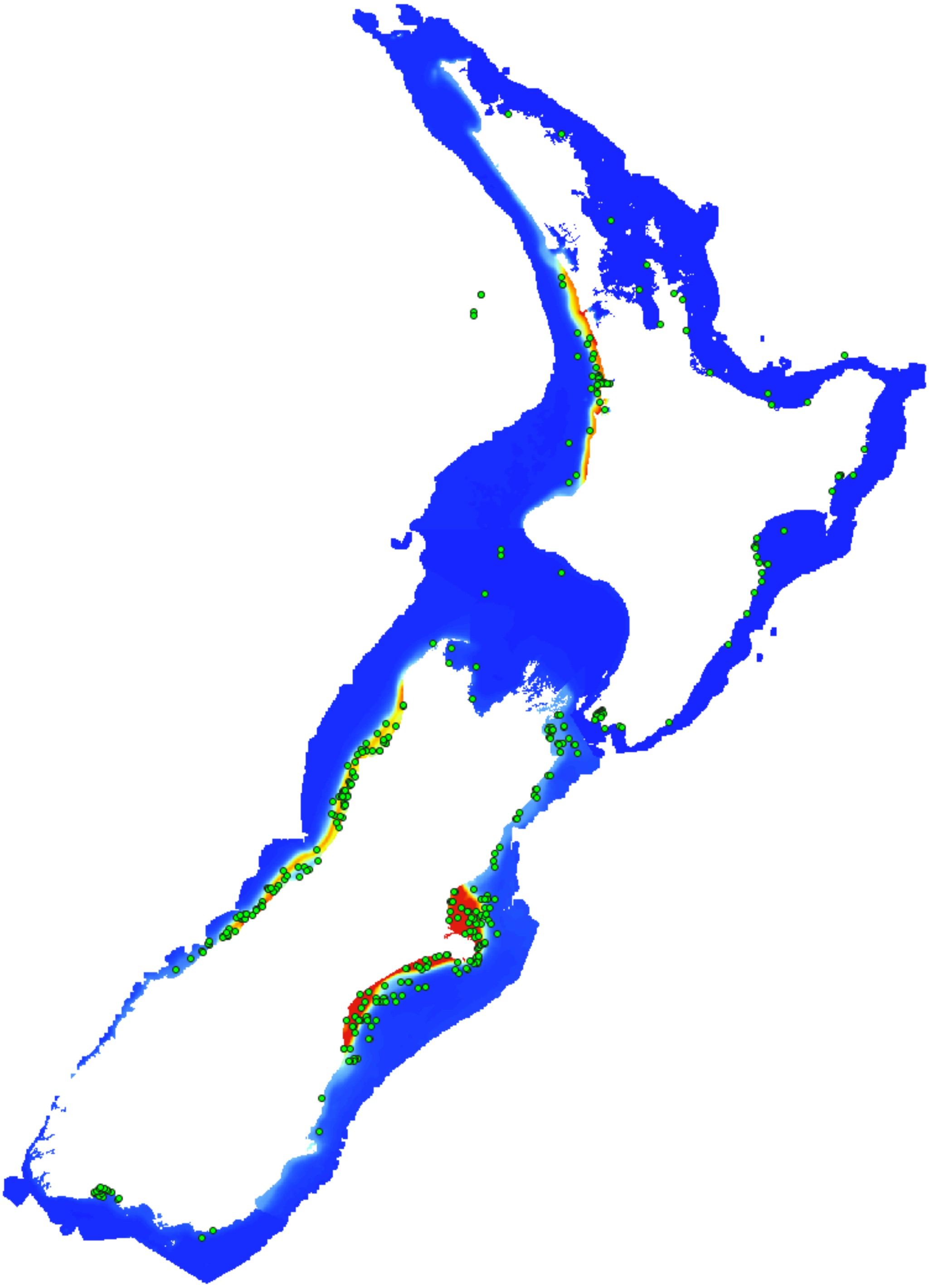

Figure 2. Roberts et al. (2019) habitat model, with Hector's and Māui dolphin sightings (green dots). All of the South Island sightings are from research surveys, and have been corrected for survey effort. The North Island sightings include public sightings that have been through a validation process. Therefore, North Island sightings (and habitat information) are not directly comparable with South Island sightings.

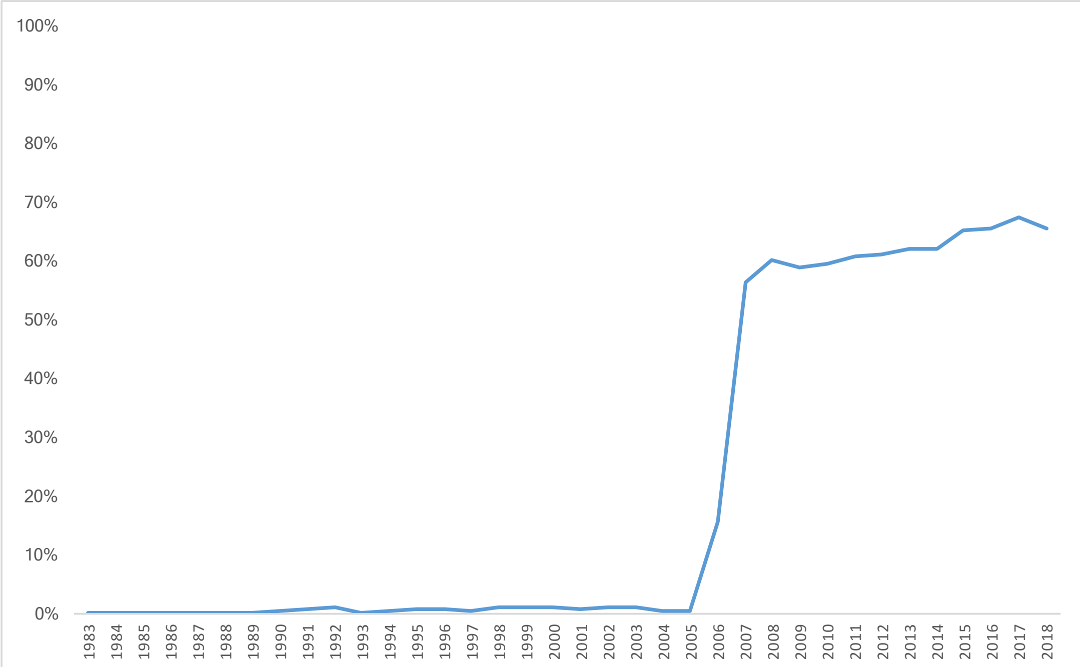

Figure 3. Proportion of gillnet fishing effort for which location is reported by latitude and longitude. Data provided by the Ministry for Primary Industries (MPI).

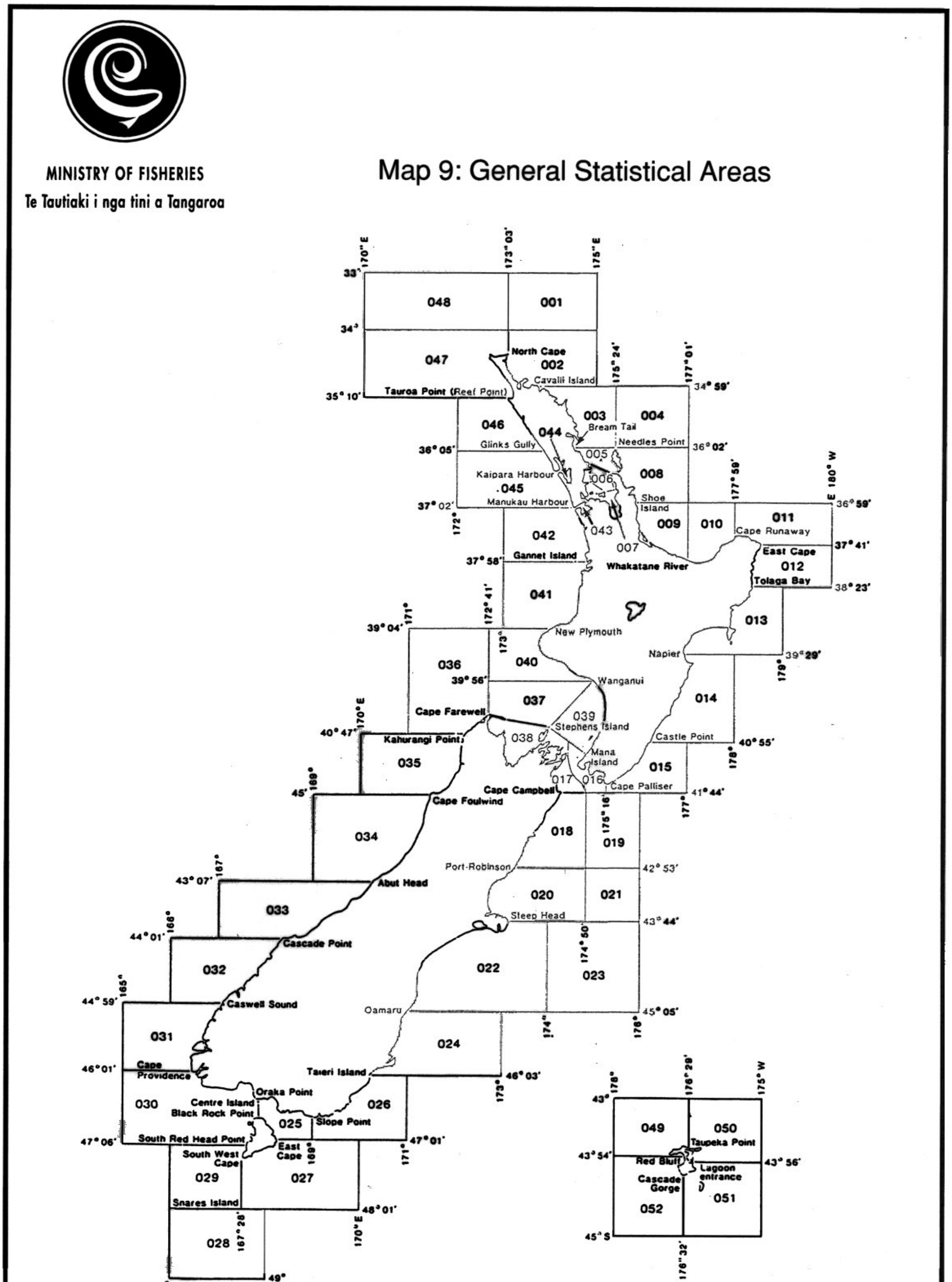

Figure 4. Fisheries statistical areas. Before 2006, the location of more than 99% of gillnet fishing effort was reported at the level of these statistical areas. In recent years, the location of about 35% of gillnet fishing effort is reported at the level of these statistical areas with the remainder reported by GPS location.

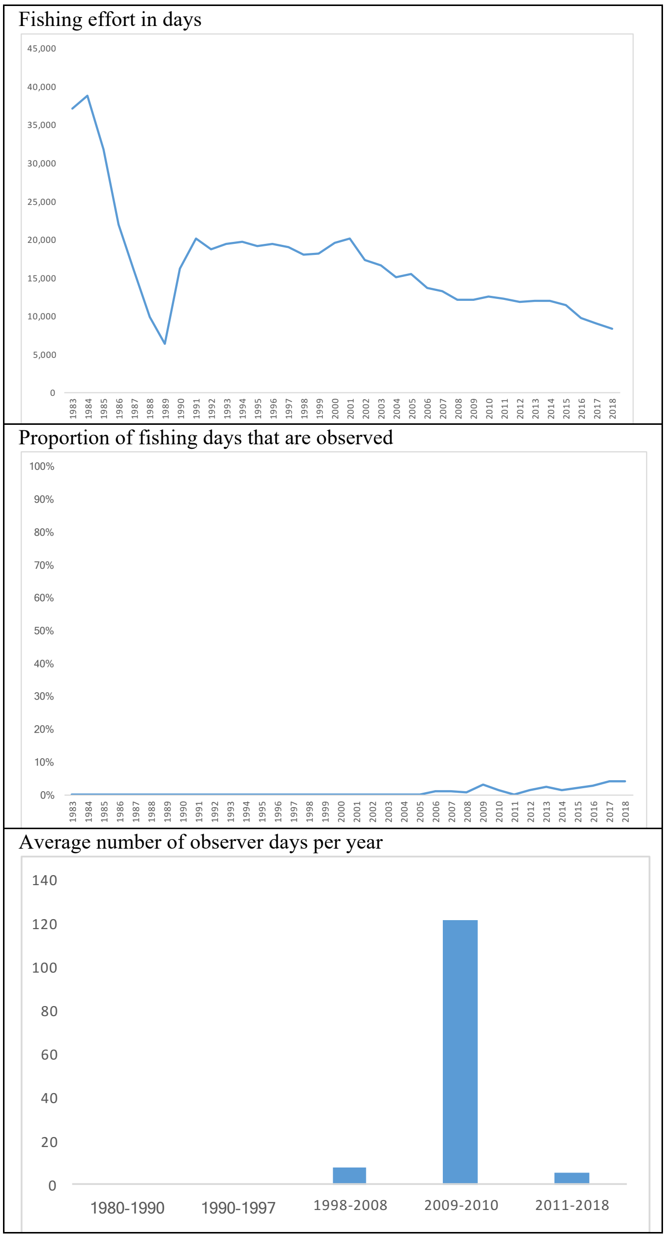

Figure 5. Gillnet fishing effort and observer coverage in Hector's and Māui dolphin habitat. Data provided by the Ministry for Primary Industries (MPI).

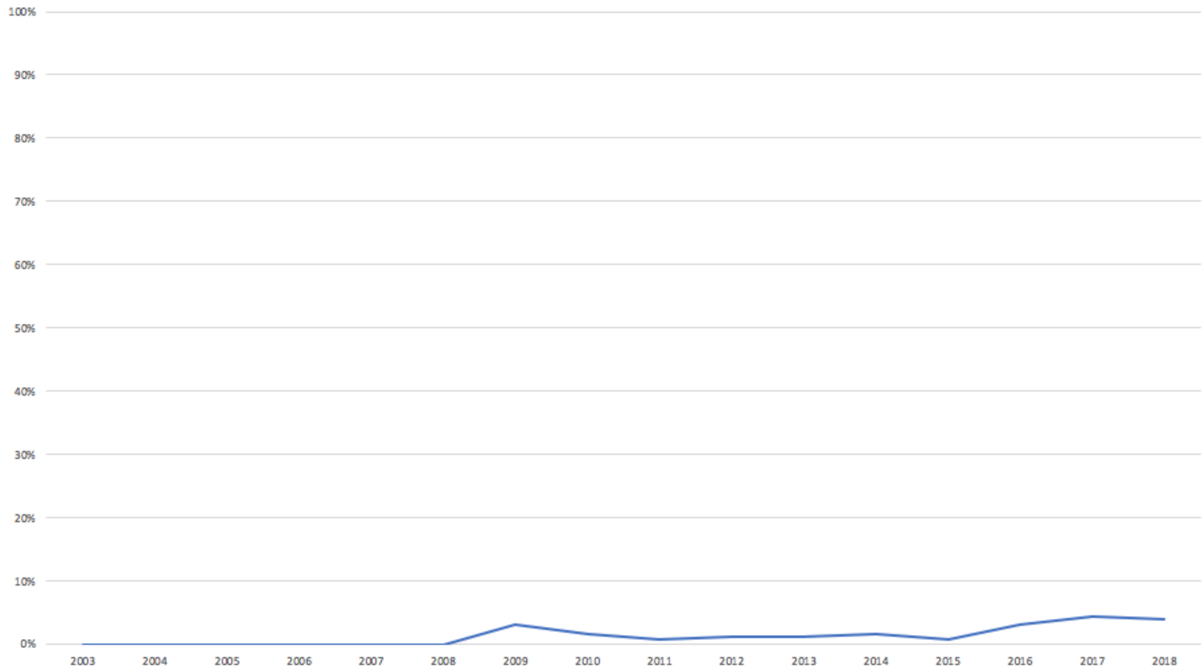

Figure 6. Proportion of observed trawling effort on inshore vessels (6-17 m in length) operating in New Zealand waters from 2003-2018. From Dragonfly (2020).

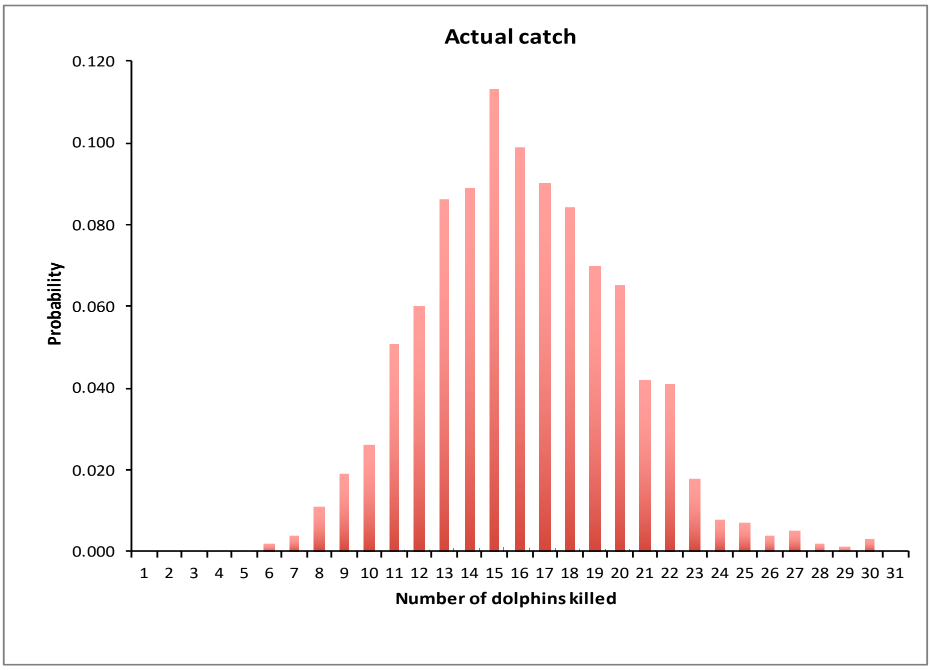

Figure 7. Bootstrap estimates of actual catch, assuming a gillnet fishery of 1000 sets/year, and a catch rate of 0.015/set. 1000 replicates.

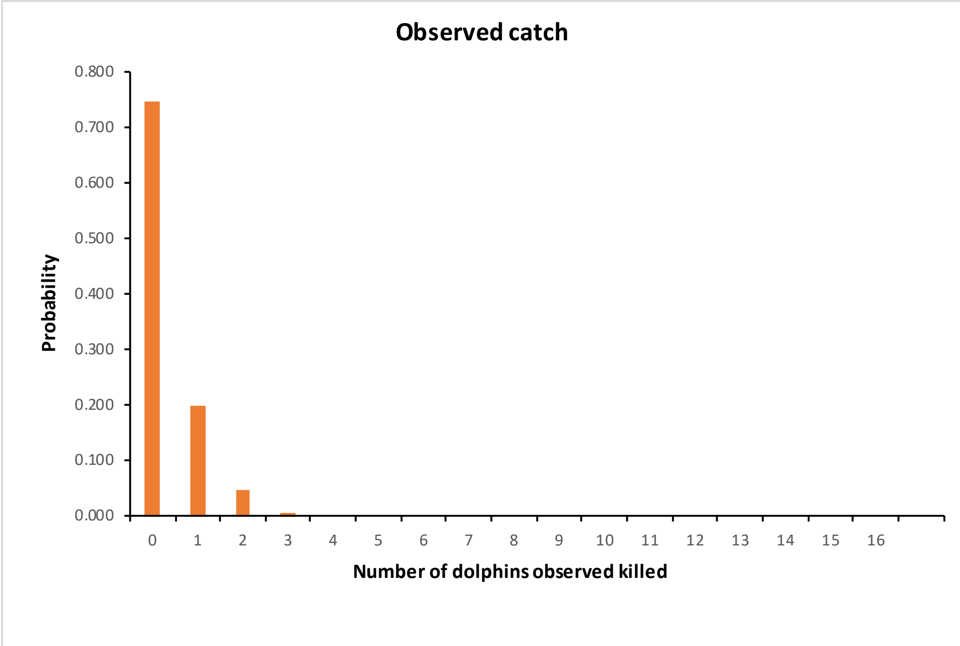

Figure 8. Annual probability of observing bycatches at 2% observer coverage, in the bootstrapping simulation of Figure 7.

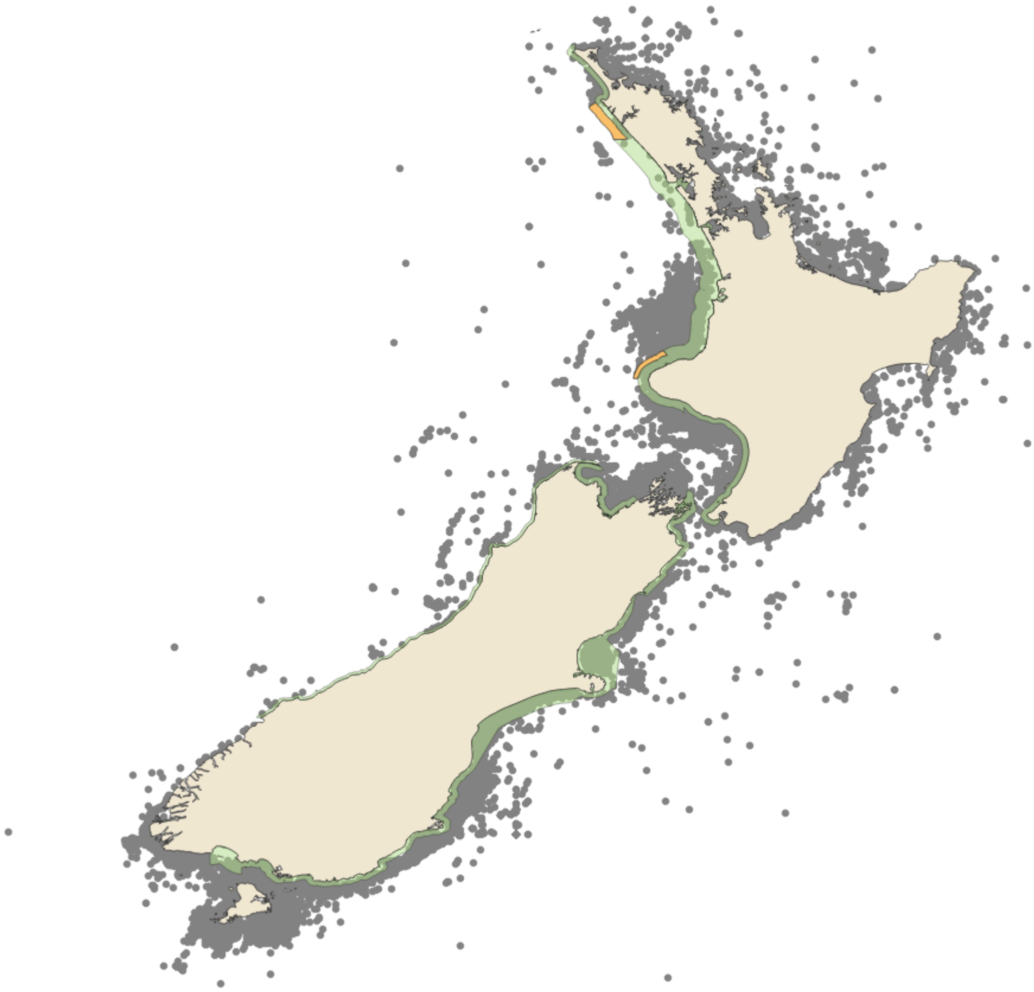

Figure 9. Gillnet fishing effort during 2009-2017 (grey dots), area closed to gillnets from 2020 (green) and area with observer coverage for Māui dolphin (orange). Fishing effort data from MPI.

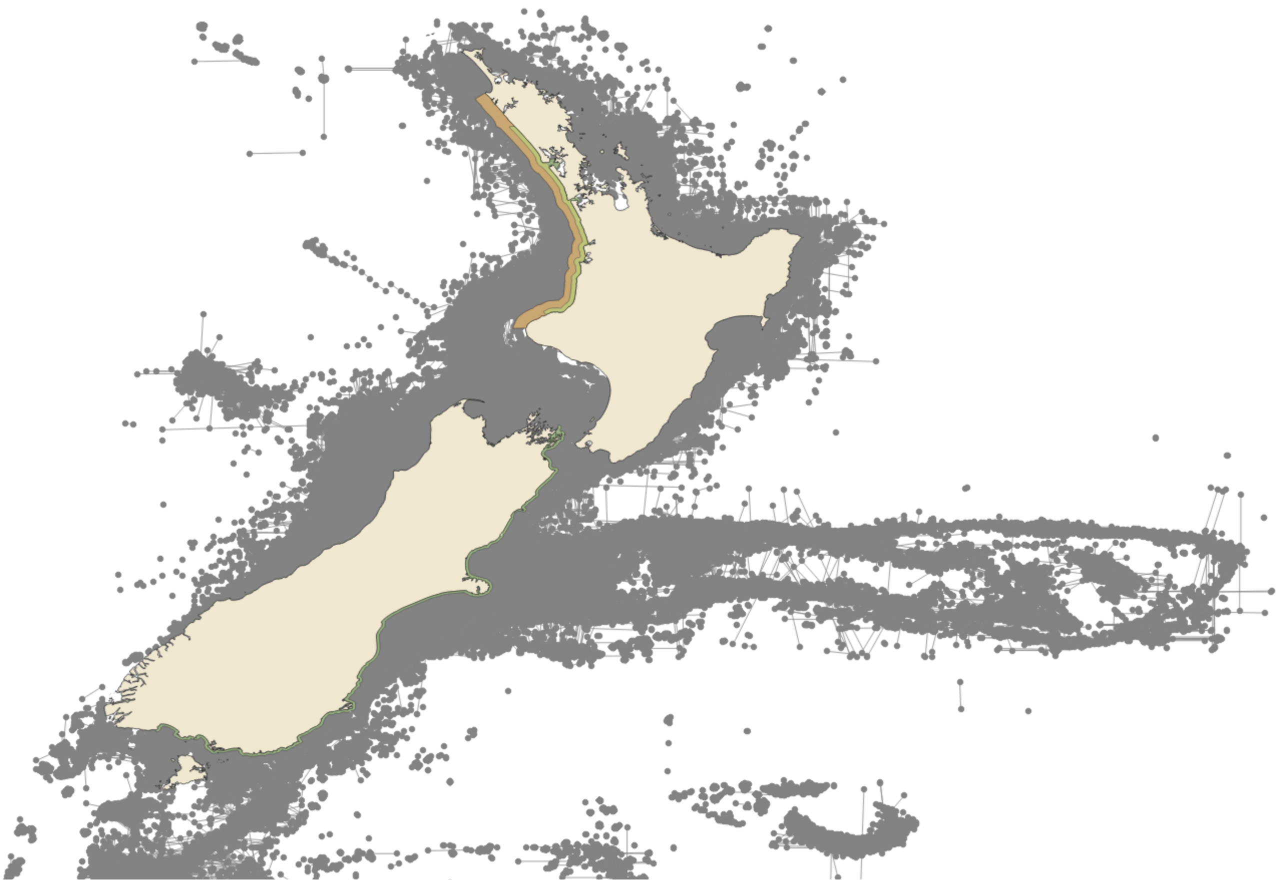

Figure 10. Trawl fishing effort during 2009-2017 (grey; data from MPI), area closed to trawling from 2020 (green) and area with Māui dolphin observer coverage (orange).

	Observer coverage				
True catch rate	1.00%	2.00%	3.00%	5.00%	10.00%
0.50%	0.951	0.907	0.865	0.775	0.602
1.00%	0.901	0.817	0.739	0.602	0.362
2.00%	0.818	0.669	0.544	0.37	0.131
3.00%	0.733	0.544	0.398	0.218	0.052
4.00%	0.659	0.436	0.298	0.13	0.016

Table 1: Annual probabilities of observing zero bycatch at given levels of observer coverage (1-10%) and true catch rate (0.4-4%), assuming fishery effort of 1000 sets/yr, monitored for 1000 years. Hence a 1% catch rate combined with 2% observer coverage results in an 81.7% chance of observing zero bycatch in that observer programme in a given year. Note that these calculations assume that bycatch events are independent. If they are not, as could be caused by dolphins living in groups affecting the likelihood of multiple captures, probabilities of observing zero bycatch would be higher than shown above.

	Estimated Bycatch	PBR	PST	PST / PBR
Hector's dolphin	59	11	74	7x
Regional populations of Hector's dolphin:				
East coast South Island	51	7	49	7x
West coast South Island	6	4	27	7x
South coast South Island	1	0.2	2	10x
North coast South Island	1	0.1	1	10x
Local populations off east coast South Island:				
Kaikoura	11	1	10	10x
Banks Peninsula	17	9	56	6x
Timaru	20	3	34	11x

Table 2. Comparison of the PBR with the PST substitute used by the New Zealand Ministry for Primary Industries (MPI) for national, regional and local populations of Hector's dolphins.