Marine reptiles – review of interactions and populations
Final Report

October 2016

Prepared for Department of Conservation

Project Code: POP2015-06

Project No: 4658

Project start date: 14 September 2015

Project end date: 20 October 2016

Final Report

Author: Dan Godoy

Karearea Consultants

Auckland, 0618, New Zealand

Abstract

Five species of marine turtles and four species of sea snakes and kraits have been recorded in New Zealand waters. These species are susceptible to adverse effects from commercial fisheries to varying degrees. This research investigated commercial bycatch data to describe the nature and extent of marine reptile interactions in New Zealand's Exclusive Economic Zone from 2008 to 2015. Existing population information was reviewed to assess potential risks to fisheries, to identify information gaps, and ultimately make recommendations to mitigate impacts. In total, 120 marine turtle bycatch records were reported while no bycatch of sea snakes or kraits were documented. Leatherback turtles (*Dermochelys coriacea*) were most frequently captured comprising 75% (*n* = 90) of all records. In contrast, green turtles (*Chelonia mydas*), hawksbill turtles (*Eretmochelys imbricata*), and loggerhead turtles (*Caretta caretta*) were captured in relatively low numbers, comprising 10% (*n* = 12), 5% (*n* = 6) and 2% (*n* = 2), respectively. The large majority of all bycatch events occurred in fisheries management areas off northeastern New Zealand (74%) and during summer (51%, *n* = 61) and autumn (38%, *n* = 45). Surface longline (SLL) activities targeting swordfish and tunas posed the greatest risk to marine turtles, recording the highest number of bycatch overall (91%, *n* = 109). In particular, leatherback turtles were most frequently captured in this fishery, accounting for 73% (*n* = 88) of total bycatch. The potentially significant threat of SLL activities to marine turtles is reflected by the annual bycatch rate (for all species combined) which, in some years, exceeded the Western and Central Pacific Fisheries Commission recommended minimal marine turtle interaction rate of 0.019 turtles per 1000 hooks. In addition, very low observer coverage was allocated to fisheries and management areas where marine turtle bycatch was most likely to occur. Overall, very little local population information is available for marine reptile species in New Zealand. Ultimately, given the potential impacts to marine turtles and information gaps identified, several recommendations are made in order to mitigate bycatch risk in New Zealand.

Keywords: New Zealand, fisheries, bycatch, marine turtle, sea snake

Contents

Introduction

Generally defined as ectotherms, marine reptiles rely on ambient temperature to regulate physiological processes necessary for reproduction and survival (Cogger 2007; Davenport 1997; Hochscheid et al. 2002). Hence, their range is normally restricted to tropical and subtropical regions of the world (Cogger 2007; Marquez 1990). Despite this, differences in their life history traits and thermal tolerances mean that some species may naturally disperse or migrate into cooler latitudes (Gaspar et al. 2012; Mrosovsky 1980). As a result, their presence in New Zealand waters varies from vagrants incidentally carried by ocean currents, seasonal visitors, to year round residents (Benson et al. 2011; Gill 1997; Godoy et al. 2011; Godoy et al. 2016). To date, five species of marine turtles and four species of sea snakes and kraits have been recorded in New Zealand waters (Gill 1997; Hitchmough et al. 2013).

Globally, marine reptiles are under serious threat due to the adverse effects of fisheries activities (Block et al. 2011; Gilman et al. 2008; Wallace et al. 2011; Wallace et al. 2013). Long-lived marine turtles are particularly vulnerable because of their highly migratory and complex life history that exposes every life stage to fisheries activities (Eckert 1995; Wallace et al. 2013). Consequently, fisheries impacts have resulted in substantial declines of marine turtle populations worldwide (Lewison & Crowder 2007; Wallace et al. 2011). Today, all marine turtle species are listed on the International Union for Conservation of Nature (IUCN) Red List of Threatened Species as Critically Endangered, Endangered, or Vulnerable (Table 1). Although sea snakes and sea kraits are also susceptible to bycatch, fisheries impacts to populations are less understood (Milton 2001). Of the four sea snake and krait species recorded in New Zealand, all are listed as Least Concern in the IUCN list.

Under New Zealand legislation, all marine reptile species are fully protected under the Wildlife Act 1953 and have been assessed according to the New Zealand Threat Classification System (NZTCS; Table 1). The Department of Conservation (DOC) is mandated to conserve and manage protected species in New Zealand. As part of this mandate the Conservation Services Programme (CSP) levies the commercial fishing industry to undertake conservation services as defined in the Fisheries Act 1996. Services include research into understanding how protected species interact and are affected by fisheries activities. To achieve this, relevant population information is also required to enable managers to develop suitable mitigation policies. Overall, local population information about these species is very limited and their interaction with fisheries is poorly understood.

Historically, the reported bycatch of marine reptiles in New Zealand fisheries has been low (Brouwer & Griggs 2009; Harley & Kendrik 2006). However, these conclusions relied exclusively on observer reports derived from the New Zealand government observer programme. Given the non-uniform distribution of observers across the New Zealand fishing fleet, observer coverage in some target fisheries (e.g. domestic shallow-set surface longline vessels) has been very low (Brouwer & Griggs 2009). As a result, the highly variable level of observer coverage has made it challenging to interpret and accurately estimate total bycatch rates and infer risks. Since 2008, commercial fishers have been required by law to report 'Non-Fish Protected Species' bycatch. Although reservations regarding the accuracy of the data supplied by commercial fishers are valid, these data may be useful in augmenting observer bycatch data to gain an understanding of protected species interaction with commercial fisheries. Therefore, this research combines observer and commercial bycatch records to gain an understanding of the nature and extent of marine reptile interactions with commercial fisheries. Commissioned under the CSP framework this population project

(POP2015-06 *Marine reptiles – review of interactions and populations*) has four main objectives:

1. To review existing information to describe the nature and extent of interactions between commercial fishing and marine reptiles.

2. To review existing information to describe population information relevant to assessing risk from commercial fishing to marine reptiles.

3. To review existing information on possible mitigation options relevant to New Zealand fisheries to minimize marine reptile bycatch.

4. To identify information gaps in the understanding of the nature and extent of interactions between commercial fishing and marine reptiles, population information and mitigation options, and provide recommendations for further research to address any gaps identified.

Table 1. List of marine reptile species recorded in New Zealand waters. International (IUCN) and National (NZTCS) Status with qualifiers and criteria included. NZTCS qualifiers: TO – Threatened Overseas; DP – Data Poor; SO – Secure Overseas.

Name and Authority	**Common name**	**NZTCS Status and Qualifiers**	**IUCN Category and Criteria**
Dermochelys coriacea (Vandelli, 1761)	Leatherback turtle	Migrant - TO	Vulnerable (globally) Critically endangered (Pacific Ocean) A2bd ver. 3.1
Chelonia mydas (Linnaeus, 1758)	Green turtle	Migrant - TO	Endangered A2bd ver. 3.1
Eretmochelys imbricata (Linnaeus, 1766)	Hawksbill turtle	Vagrant – TO	Critically Endangered A2bd ver. 3.1
Caretta caretta (Linnaeus, 1758)	Loggerhead turtle	Vagrant – TO	Vulnerable A2b ver. 3.1
Lepidochelys olivacea (Eschscholtz, 1829)	Olive Ridley turtle	Vagrant – TO	Vulnerable A2bd ver. 3.1
Pelamis platura (Linnaeus, 1766)	Yellow-bellied sea snake	Not Threatened – DP, SO	Least Concern ver. 3.1
Laticauda colubrina (Schneider, 1799)	Yellow-lipped sea krait	Vagrant – SO	Least Concern ver. 3.1
Laticauda saintgironsi (Cogger & Heatwole, 2005)	Saint-Girons' sea krait	Vagrant – SO	Least Concern ver. 3.1
Laticauda laticaudata (Linnaeus, 1758)	Blue-lipped sea krait	Vagrant – SO	Least Concern ver. 3.1

Methods

Information regarding the nature and extent of interactions between marine reptiles and commercial fisheries in New Zealand waters was obtained from five main sources. They include published and unpublished literature, the commercial catch database (*warehou*), the central observer database (COD), the DOC herpetofauna database, and the New Zealand marine turtle sighting and stranding database (research database curated by D Godoy). Data was cross-referenced between all sources to ensure duplicates were omitted. Where duplicates occurred, information from Observer (CSP) and Commercial Non-Fish Protected Species Bycatch (NFPS) records were combined into single records. For the purpose of this report, the identification of reported species has been assumed to be correct unless additional information (e.g. photographic evidence) confirmed otherwise.

Relevant Fisheries Management Area (FMA), Statistical Area (SA) and bathymetry map layers were obtained for mapping and analysis. Bycatch data was analysed and distribution maps produced to highlight FMA and fishing methods most at risk of bycatch. Records were omitted from the analysis that did not include a latitude/ longitude, Fishery Statistical Area, or Fisheries Management Area given that a bycatch location could not be estimated from the data. Records where a latitude/ longitude was not recorded but SA or FMA was reported, the area centroid was used to position bycatch events. In addition, consideration of the target species and fishing method was also used to further inform positioning of bycatch events in cases where only SA or FMA was provided. Where latitude/ longitude was not specifically recorded for a bycatch event, the Catch Effort start position was used to locate the bycatch event.

Bycatch for each marine reptile species was summarised and tabulated according to fishing year (defined as the period 1 July – 30 June), FMA, fishing method and target species. Bycatch rates (catch by unit effort) were calculated for all species combined and categorised by fishing year and FMA. Average annual bycatch was calculated for each species however data from the 2015/16 fishing year was excluded because this period only encompassed five months (1 July 2015 – 30 November 2015). At risk FMAs, target fishery, fishing method, geographic location, and time period was identified for each species. Published and unpublished population information for each marine reptile species found in New Zealand waters was reviewed and information gaps were identified. Fisheries risk, mitigation options and future research recommendations have been made.

Results

Extent of interactions

In total, 120 marine reptile bycatch records were documented from 1 July 2008 to 30 November 2015, excluding two records reported from outside New Zealand's EEZ (i.e. from the tropical Pacific), and four duplicate records that were combined into single records. All bycatch records were of marine turtles, while no incidences of sea snake or sea krait bycatch were reported. Leatherback turtles were the most frequently captured species comprising 75% (*n* = 90) of all reported events, followed by green turtles 10% (*n* = 12), hawksbill turtles 5% (*n* = 6), and loggerhead turtles 2% (*n* = 2) (Table 2). Unidentified marine turtles accounted for 8% (*n* = 10) of all records. Although olive ridley turtles (*Lepidochelys olivacea*) have been recorded in New Zealand waters (comprising of strandings, sightings, recreational incidental captures; D. Godoy, unpubl. data) there were no records of commercial bycatch. Inter-annual total bycatch (all records combined) varied considerably from 2 (2015/16: encompassing only 5 months) to 28 (2012/13), resulting in an average of 17 bycatch events each fishing year (S.D. 9.4, *n* = 118). For all records combined, 90% (*n* = 106) were from the North Island region with the majority of bycatch events occurring off eastern North Island in FMA 1 (55%, *n* = 66) and FMA 2 (19%, *n* = 23) (Table 2, Fig. 1). Only 12 (*n* = 10%) records occurred in regions off the South Island, with all from FMA 7 (Fig. 1). Most bycatch occurred during summer (51%, *n* = 61) and autumn (38%, *n* = 45) when sea surface temperatures were between 22.8 °C (March) and 15.1 °C (June).

Surface long line (SLL) activities targeting swordfish and tunas captured the highest number of marine turtles (91%, *n* = 109) with leatherback turtles accounting for 73% (*n* = 88) overall (Table 4, Fig. 2). SLL activities were the only fishing method that resulted in more than one turtle capture (2-3 turtles) in an individual fishing event (e.g. net, tow, line set etc). Five multiple capture events were reported comprising three in FMA 1; two in FMA 2; and one in FMA 7; indicating a degree of spatio-temporal clustering protected species. All were leatherback turtles except the capture of two unidentified turtles in FMA 7. Captures were also notably prevalent in oceanic waters beyond the continental shelf (>200 m) where 92% (*n* = 110) of all marine turtle bycatch occurred in this habitat (Fig. 1). This is particularly relevant to leatherback turtles where 98% (*n* = 88) of all bycatch records of this species occurred in oceanic habitats (Fig. 2). In contrast, green turtles were most often captured in neritic habitats i.e. over the continental shelf in water depths < 200 m (Fig. 3) and mostly in fishing methods other than SLL i.e. Bottom longline (2), Bottom trawl (3), Set net (1), and Trawl (1) (Table 4). Similarly to leatherback records, all records of hawksbill bycatch (6) occurred in oceanic habitats within FMA 1 and FMA 9 (Fig. 4). However, it should be noted that five of six hawksbill records were reported by the same vessel (vessel key: 8075) between February and August 2012. Given that these records were not confirmed by a CSP Observer or through photographic images, suggests some speculation to the validity of this species' identification. Only two records of loggerhead turtles were reported between 1 July

2008 and 30 November 2015, with a single bycatch event in SSL and Bottom trawl (BT) activities in oceanic and neritic waters, respectively (Fig. 5). All unidentified turtles were captured in oceanic habitats, and with the exception of a single capture in a BT event, all were captured in SLL activities targeting tunas (Fig. 6). All captured marine turtles, except one leatherback captured in a set net, were alive when found (Table 5). Of all turtles captured alive, 88% (n = 106) were recorded as uninjured regardless of fishing method, however, 11% (n = 13) were reported as sustaining injuries. Where injuries were described or coded, most (6) were due to hook injuries sustained in SLL activities. When described, turtles were either hooked in the mouth (1 unidentified turtle) or flipper (4 leatherback turtles and 1 unidentified turtle). In addition, records state the all injured turtles were released alive, yet the snood was cut and therefore the turtle swam away with the hook and snood still attached.

Table 2. Marine reptile bycatch from 1 July 2008 to 30 November 2015 (*n* = 120) by Fisheries Management Area (FMA). Data includes all commercial and observer records across all fishing methods and target species.

FMA		Leatherback turtle	Green turtle	Hawksbill turtle	Loggerhead turtle	Unidentified turtle	Total
1	Auckland (East)	52	7	1	1	5	66
2	Central (East)	22	1				20
3	South-East (Coast)						0
4	South-East (Chatham Rise)						0
5	Southland						0
6	Sub-Antarctic						0
7	Challenger/Central (Plateau)	8	1			2	11
8	Central (Egmont)	1					1
9	Auckland (West)	7	3	5	1	3	19
10	Kermadec						0
Total		**90**	**12**	**6**	**2**	**10**	**120**

Table 3. Annual marine reptile bycatch from 1 July 2008 to 30 November 2015 (*n* = 120). Data includes all commercial and observer records across all fishing methods and target species.

Fishing year	Leatherback turtle	Green turtle	Hawksbill turtle	Loggerhead turtle	Unidentified turtle	Total
2008/2009	7	3				10
2009/2010	2	1			2	5
2010/2011	17	2	1	1	4	25
2011/2012	18	1	2			21
2012/2013	21	1	3		3	28
2013/2014	7	2		1	1	11
2014/2015	17	1				18
2015/2016[¥]	1	1				2
Total	**90**	**12**	**6**	**2**	**10**	**120**

¥ Fishing year 2015/16 only covers 5 months (1 July 2015 – 30 November 2015).

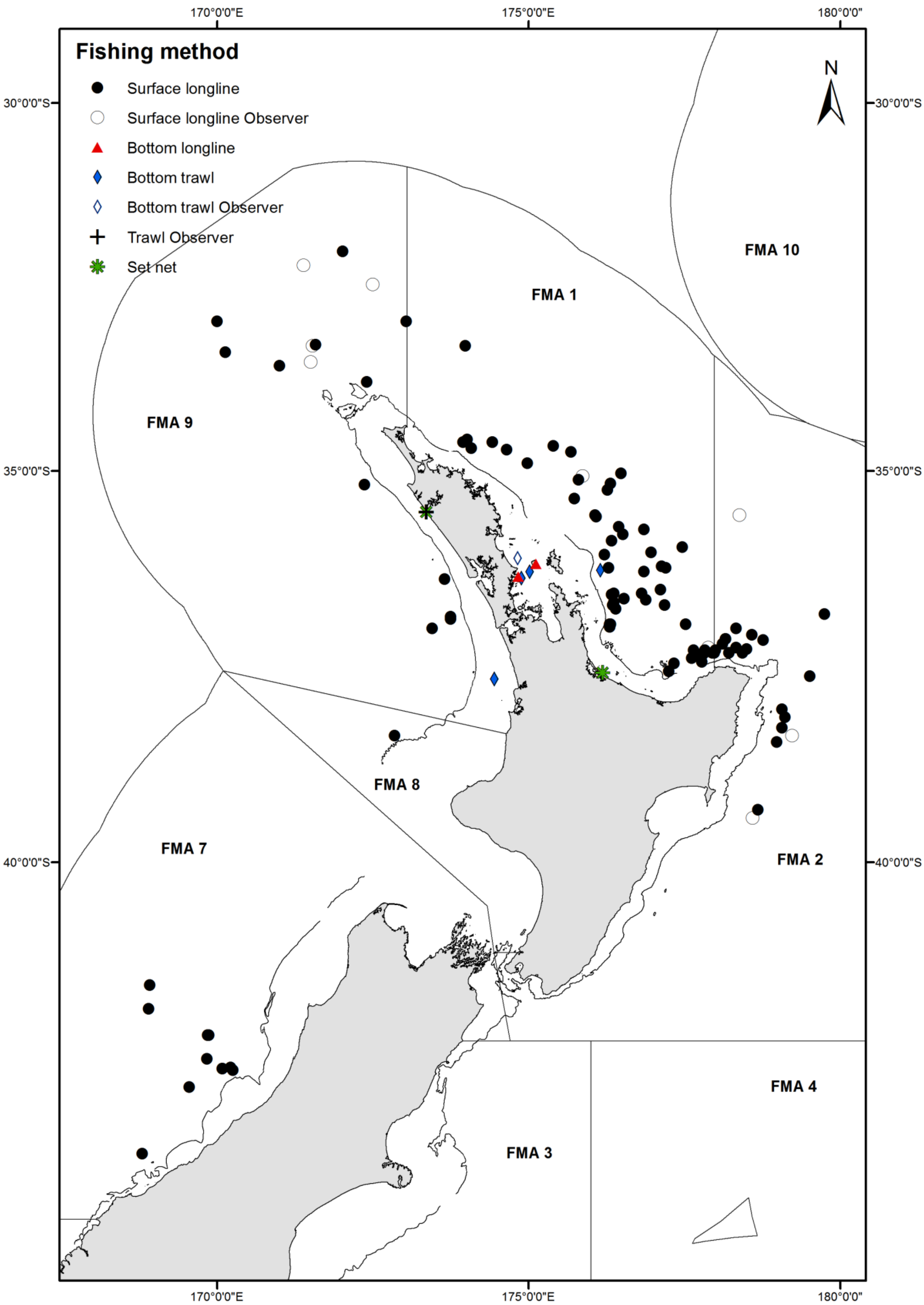

Figure 1. Distribution of all reported marine turtle bycatch data from 1 July 2008 to 30 November 2015 (*n* = 120). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

Table 4. Marine reptile bycatch data from 1 July 2008 to 30 November 2015 (n = 120). Data includes all commercial and observer records across all fishing methods and target species.

Fishing method Target species	**Leatherback turtle**	**Green turtle**	**Hawksbill turtle**	**Loggerhead turtle**	**Unidentified turtle**	**Total**
Bottom longline		**2**				**2**
Snapper		2				2
Bottom trawl	**1**	**3**		**1**	**1**	**6**
John dory		1				1
Scampi					1	1
Snapper		1				1
Tarakihi	1					1
Trevally				1		1
Unknown		1				1
Set net	**1**	**1**				**2**
Flatfish	1					1
Grey mullet		1				1
Surface longline	**88**	**5**	**6**	**1**	**9**	**109**
Bigeye tuna	48	2	4	1	6	61
Southern bluefin tuna	10	2			3	15
Swordfish	26	1	2			29
Pacific bluefin tuna	1					1
Unknown	3					3
Trawling		**1**				**1**
Trevally		1				1
Total	**90**	**12**	**6**	**2**	**10**	**120**

Table 5. Marine reptile bycatch data from 1 July 2008 to 30 November 2015 (n = 120) and capture status (alive – uninjured, alive – Injured, dead). Data includes all commercial and observer records across all fishing methods.

Species Fishing method	Captured alive – uninjured	Captured alive – injured	Captured dead	Total
Green turtle	**12**			**12**
Bottom longline	2			2
Bottom trawl	3			3
Surface longline	5			5
Set net	1			1
Trawling	1			1
Hawksbill turtle	**5**	**1**		**6**
Surface longline	5	1		6
Leatherback turtle	**78**	**11**	**1**	**90**
Bottom trawl	1			1
Surface longline	77	11		88
Set net			1	1
Loggerhead turtle	**2**			**2**
Bottom trawl	1			1
Surface longline	1			1
Unidentified turtle	**9**	**1**		**10**
Bottom trawl		1		1
Surface longline	9			9
Total	**106**	**14**	**1**	**120**

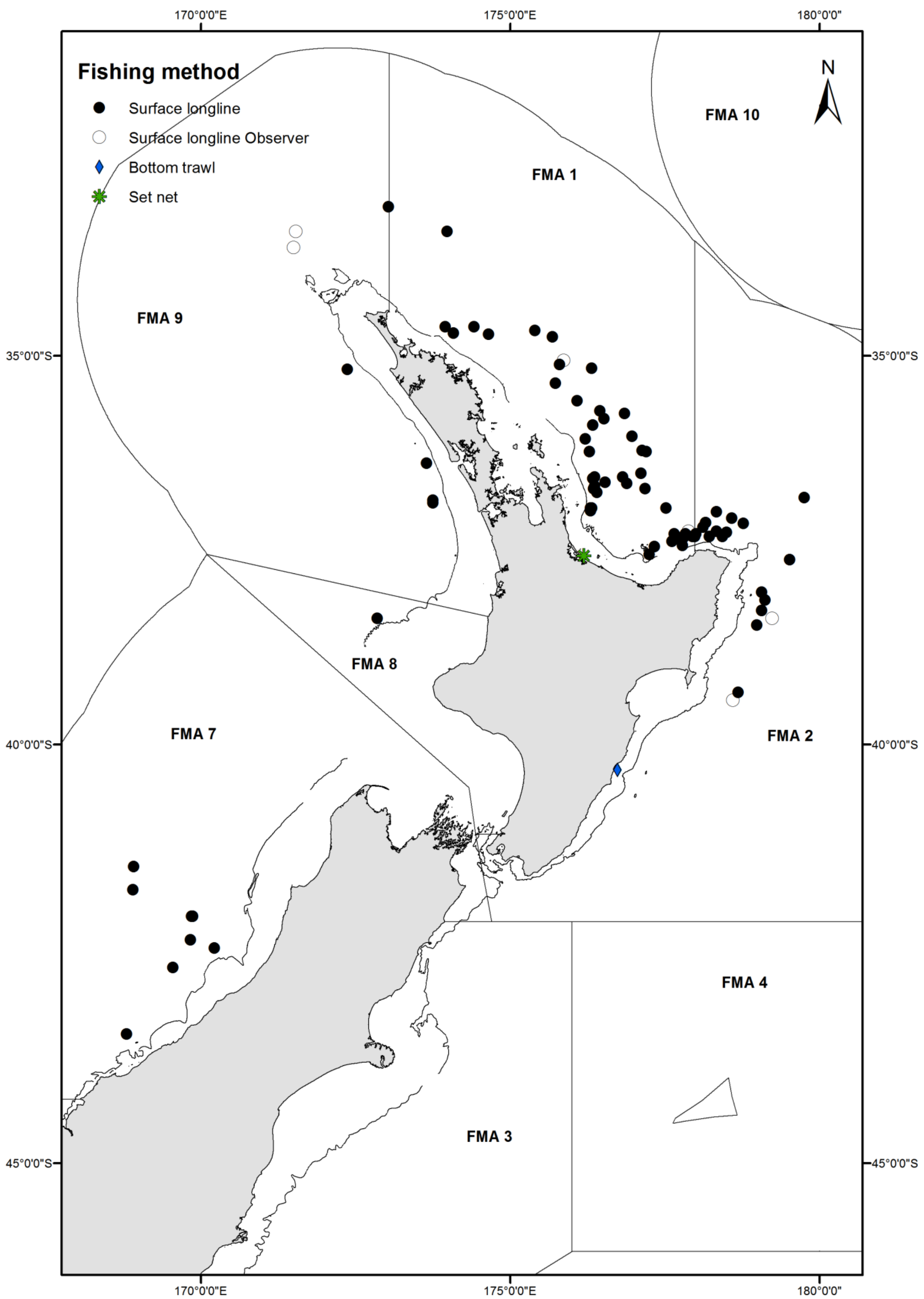

Figure 2. Distribution of reported leatherback turtle bycatch from 1 July 2008 to 30 November 2015 (n = 90). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

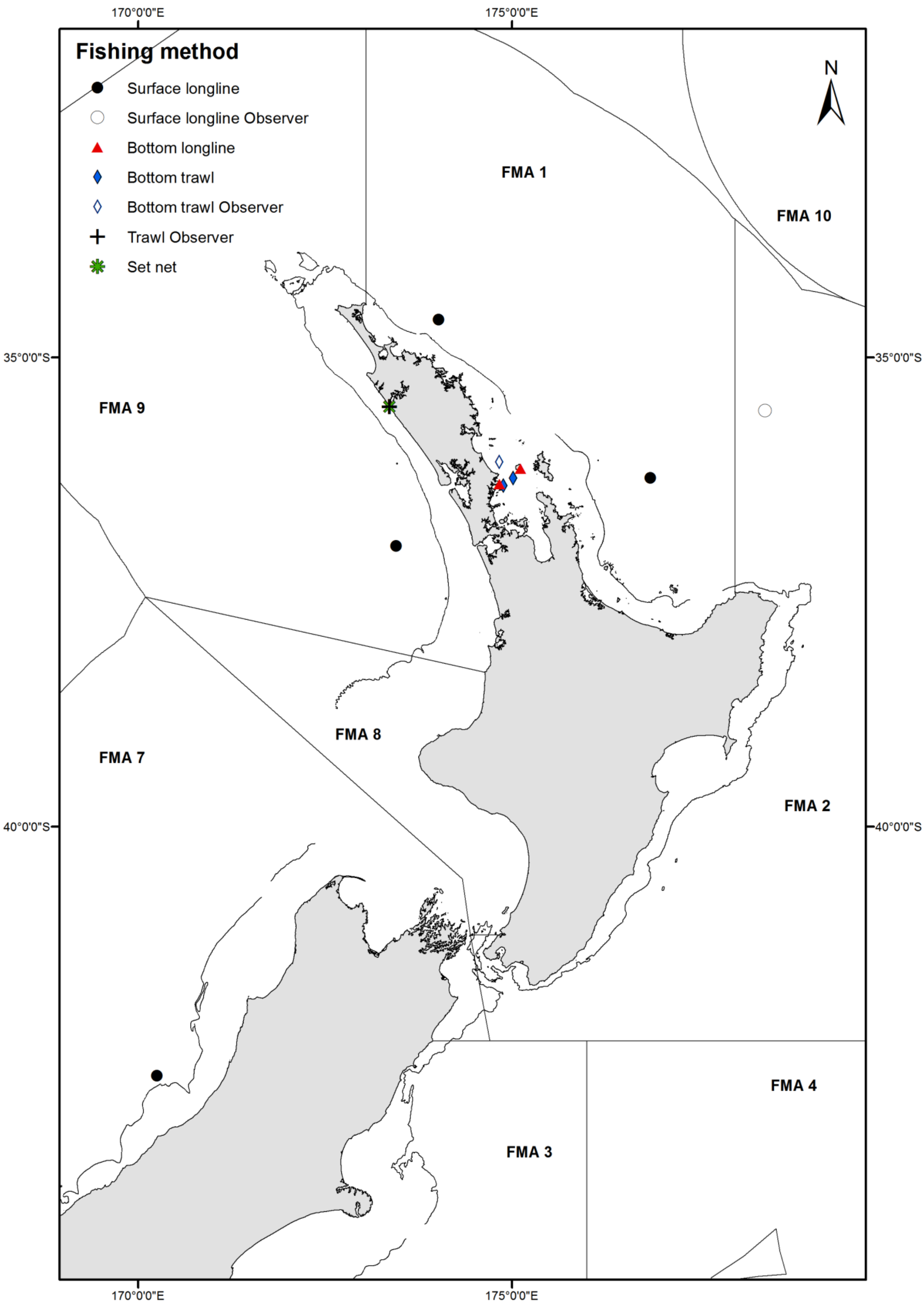

Figure 3. Distribution of reported green turtle bycatch from 1 July 2008 to 30 November 2015 (n = 12). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

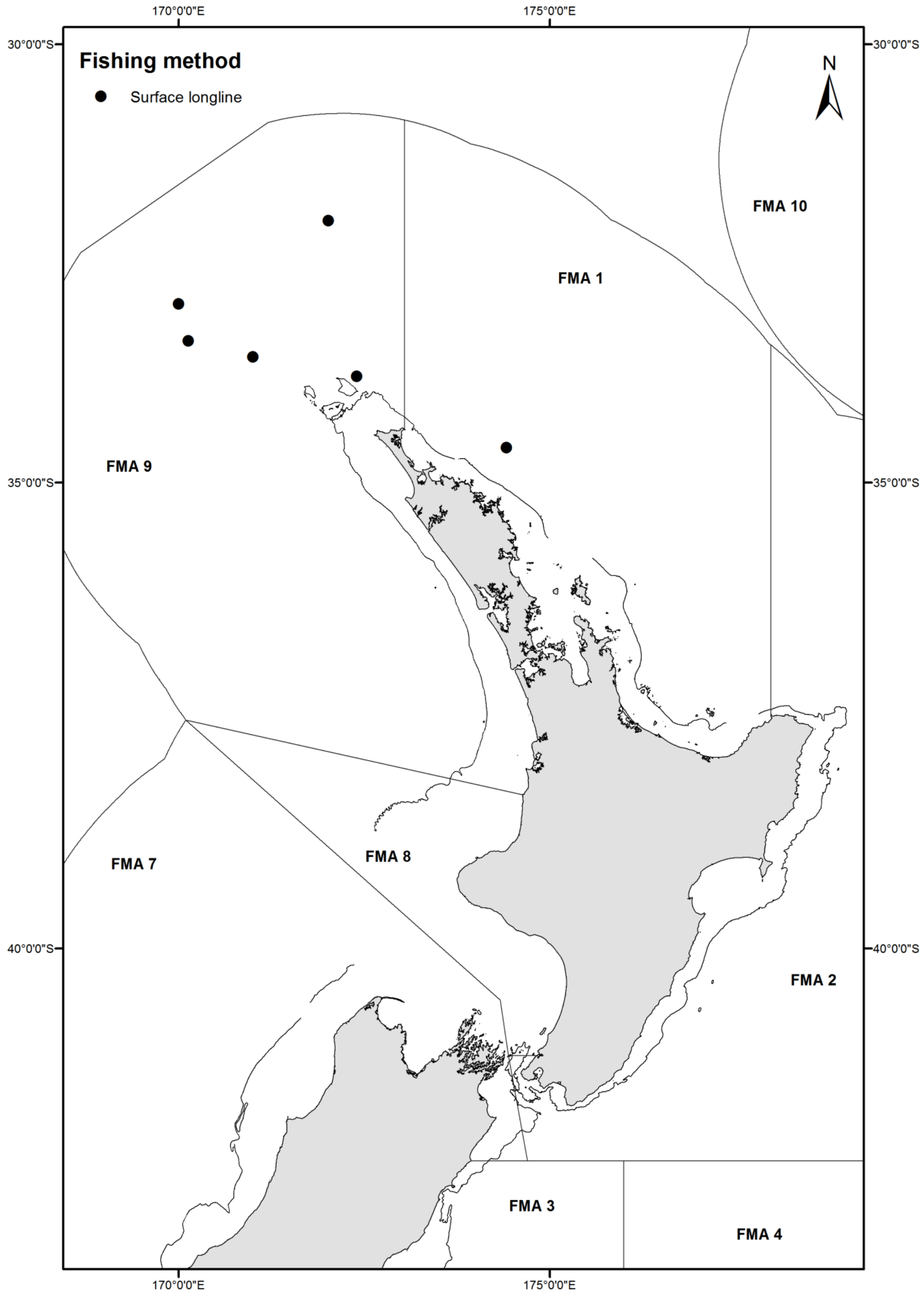

Figure 4. Distribution of reported hawksbill turtle bycatch from 1 July 2008 to 30 November 2015 (n = 6). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

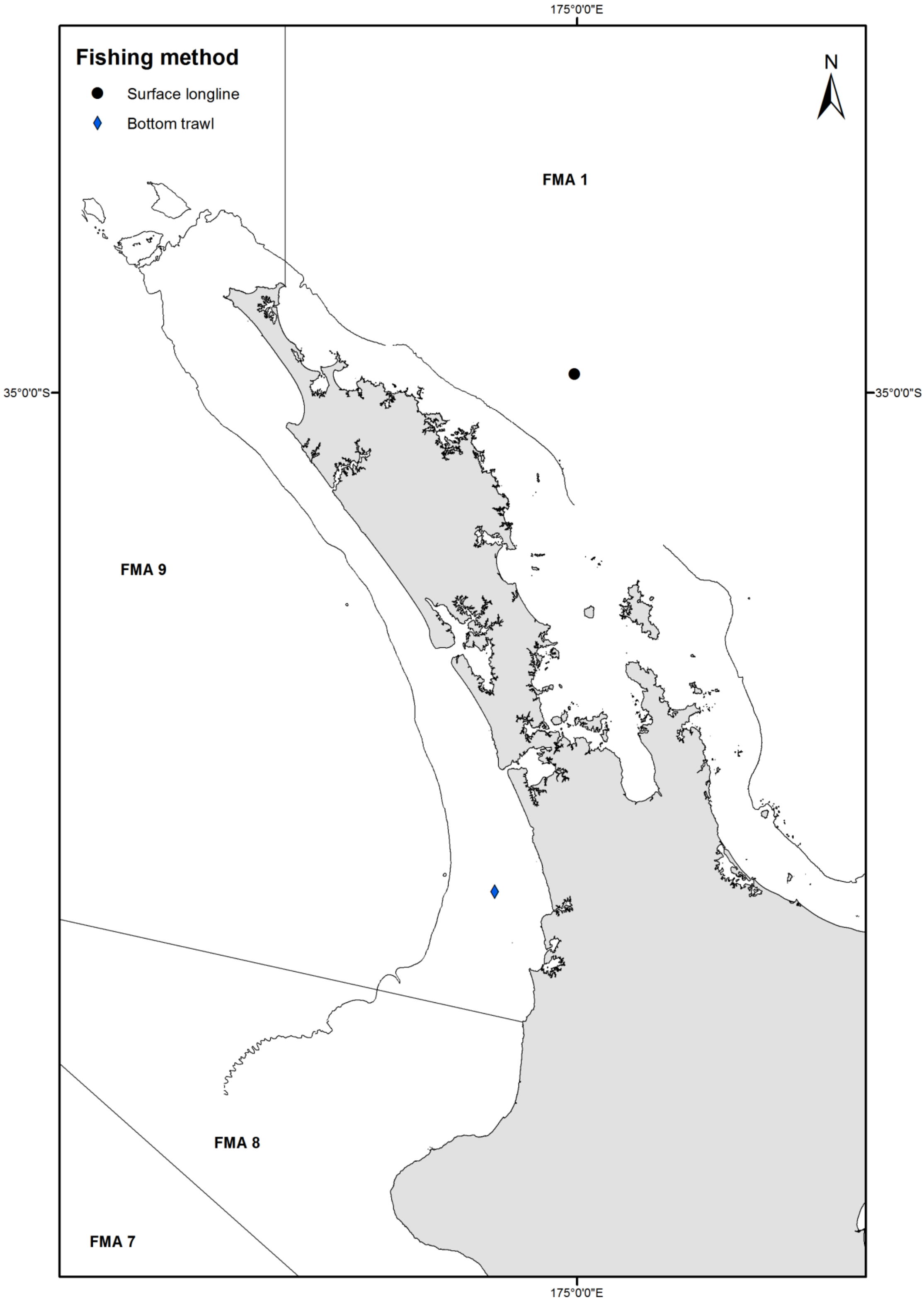

Figure 5. Distribution of reported loggerhead turtle bycatch from 1 July 2008 to 30 November 2015 (*n* = 2). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

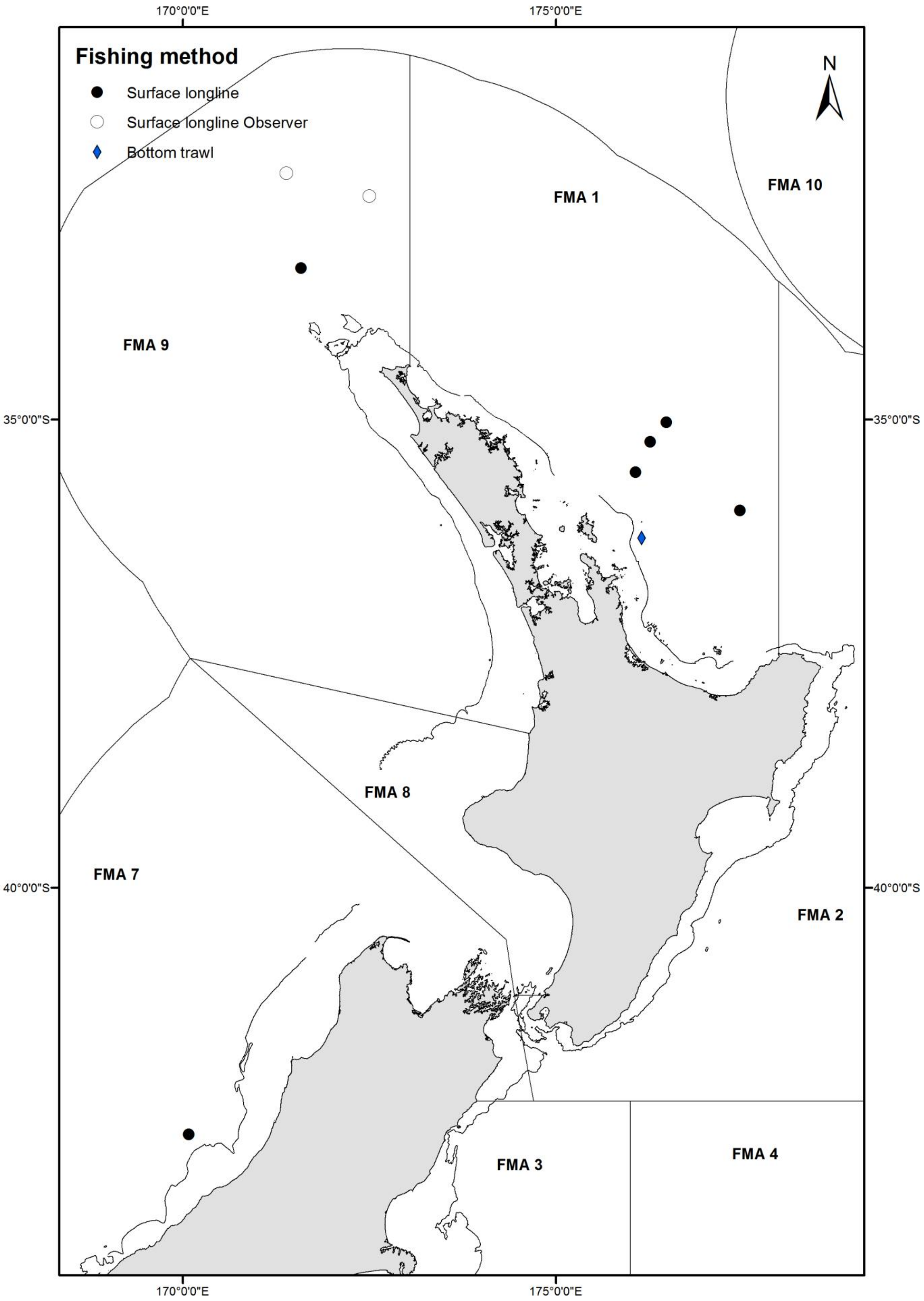

Figure 6. Distribution of reported unidentified turtle bycatch from 1 July 2008 to 30 November 2015 (n = 10). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries are shown.

Observer coverage and capture rate

An assessment of observer coverage across all species, FMA and fishing method show that CSP observed bycatch events accounted for only 9% (*n* = 11) of bycatch reports while non-observed vessels accounted for 91% (*n* = 109) of all records. Specifically, in the domestic surface longline fleet the average observer coverage was very low (5.8%) yet accounted for the highest marine turtle captures overall (106; Table 6). In contrast, observer coverage in the foreign charter surface longline fleet averaged 99.5% coverage, yet only three turtles were incidentally captured by these vessels over the same period. This highlights that observer effort is not allocated where most bycatch is likely to occur, particularly in the domestic surface longline fishery targeting bigeye tuna, southern bluefin tuna and swordfish. In addition, the annual SLL (domestic and foreign charter; Appendix 1) bycatch rates in certain FMA exceeded the WCPFC recommended minimal marine turtle interaction rate of 0.019 turtles per 1000 hooks for shallow-set longline fisheries (Brouwer & Bertram 2009). In particular, bycatch rates in FMA 1 (2009/10, 2010/11, 2011/12, 2012/13, 2014/15), FMA 2 (2012/13), and FMA 9 (2011/12, 2013/14) frequently exceeded these limits in at least one target fishery (Appendix). The high bycatch rate of 0.0849 in FMA 8 during 2012/13 should be considered with caution given that the rate is based on a single capture and low SLL fishing effort during that season.

Table 6. Surface longline fishing effort (hooks set) for each target species by the domestic and foreign charter fleets, observed effort and total marine reptile captures from 1 July 2008 to 30 November 2015 (*n* = 109). Data includes all commercial and observer records.

Target species	Domestic hooks set	Observed hooks	% observed	Bycatch no.	Charter hooks set	Observed hooks	% observed	Bycatch no.
Bigeye tuna	8,012,139	343,013	4.3	63	56,350	56,350	100	1
Southern bluefin tuna	6,908,081	520,052	7.5	13	4,049,398	4,004,912	98.9	2
Swordfish	1,527,353	101,778	6.7	29				
Pacific bluefin tuna	134,553	0	0.0	1				
Unknown	1,000	0	0.0	0				
Total	**16,582,126**	**964,843**	**5.8**[¥]	**106**	**4,105,748**	**4,061,932**	**99.5**[¥]	**3**

¥ Average percentage of hooks observed.

Population information and data gaps

Sea snakes and kraits

Sea snake and sea krait species are variably distributed throughout the tropical Pacific and Indian Oceans (Dunson 1975). Nineteen species are found in Oceania with four species (representing two genera) recorded in New Zealand waters (Polidoro et al. 2011; Gill 1997; Hitchmough et al. 2013; McCann 1966b). Ocean currents occasionally disperse these species into temperate waters, including New Zealand where they are occasionally found stranded ashore, predominantly in Northland (Gill 1997; McCann 1966b). The only representative of the genus *Pelamis* found in New Zealand, the yellow-bellied sea snake, *P. platura*, is the most pelagic of all sea snakes and is widely distributed across tropical Pacific and Indian Oceans (Dunson and Ehlert 1971, Graham et al. 1971). This species is viviparous and completes its life cycle at sea (Cogger 2007). No records of commercial bycatch have been reported for this species in New Zealand waters. Current knowledge of this species in New

Zealand is very poor. Gill (1997) found all stranded specimens found in New Zealand fell within the size range for adults, however no information exists in terms of ecology, regional connectivity or genetic origin. Across its entire range this population is considered stable and is listed as Least Concern in the International Union for Conservation of Nature (IUCN) Red List (Guinea et al. 2010). Threats to this species from fisheries activities is limited and poorly understood however minor threats may include bycatch in squid fisheries (Guinea et al. 2010; Polidoro et al. 2011).

The three sea krait species of the genera *Laticauda* (*L. colubrina, L. saintgironsi, L. laticaudata*) recorded in New Zealand are normally distributed throughout the tropical western Pacific Ocean (Cogger & Heatwole 2006; Gill 1997). These semi-aquatic oviparous marine snakes are mostly reef dwelling and retain a reproductive link to terrestrial habitats (Cogger 2007; Cogger & Heatwole 2006). No records of commercial bycatch have been reported for this species in New Zealand waters. Current knowledge of this species in New Zealand is very poor, yet those found stranded in New Zealand are most likely vagrants incidentally carried by ocean currents from tropical regions (Gill 1997). Across their range these species are considered stable and are listed as Least Concern in the IUCN Red List. Threats to these species from fisheries activities is not well understood however they can be impacted by trawling activities over continental shelf habitats (Cogger 2007; Polidoro et al. 2011). Overall, the risk to sea snakes and sea kraits from fisheries activities in New Zealand is considered low given that a) these species are primarily tropical and low numbers of have been recorded in New Zealand, b) no records of commercial bycatch have been identified in this review, and c) their populations are stable overseas.

Leatherback turtle (Dermochelys coriacea)

The leatherback turtle is the only extant representative of the family Dermochelyidae and is morphologically distinct from all other marine turtles (family: Cheloniidae) (Pritchard & Mortimer 1999; Pritchard 1997). Unlike Cheloniid marine turtles, the leatherback's carapace lacks any keratinised external scutes, instead having seven longitudinal ridges covered with a leathery skin (Pritchard & Mortimer 1999). It is also the largest marine turtle species, with adults attaining more than 2 m in total length and weighing an excess of 500 kg (Eckert et al. 2012). Its large size, unique morphology and coloration (black dorsally with white spots) make this species easily distinguished from all other marine turtle species.

The leatherback is the most widely distributed of all marine turtle species, ranging circum-globally throughout pelagic and neritic waters of tropical and temperate regions (Eckert et al. 2012; Benson et al. 2011). Having endothermic characteristics, adults of this species frequently undertake extensive seasonal foraging migrations into highly productive cold-temperate waters, feeding primarily in the epi-pelagic zone on gelatinous zooplankton (James et al. 2005; Davenport 1997; Davenport 1998; Benson et al. 2007; Saba et al. 2008). Consequently, they have been reported as far north as Norway (c. 71^{o} N) and as far south as New Zealand (Foveaux Strait: c. 47^{o} S) (Carriol & Vader 2002; Eggleston 1971). Despite its extensive biogeographical range, nesting is primarily restricted within tropical latitudes with nesting populations strongly subdivided between and within ocean basins (Eckert et al. 2012). No nesting occurs in New Zealand (Gill 1997; D. Godoy pers. obs.). Neonate hatchlings disperse into oceanic habitats, yet nothing is known of post-hatchling dispersal in the open ocean (Eckert et al. 2012). Juveniles will remain in warm oceanic habitats (>26 ^{o} C) until reaching maturity at > 120 cm curved carapace length (CCL) (Eckert et al. 2012). Sexually mature adults may expand their range into temperate zones as their thermal tolerance increases. Mature adults of both sexes will migrate to natal nesting areas for the remainder of their reproductive life. Age at maturity is estimated at 24.5-29 years with a generation length of approximately 30 years (Avens et al. 2009; Eckert et al. 2012).

The global population exhibits shallow phylogenetic structuring and comprises seven genetically distinct subpopulations (Dutton et al. 1999; Wallace et al. 2010). Seven geographically and demographically distinct regional management units (RMU) have been defined (Wallace et al. 2010). In the Pacific region, two distinct subpopulations (RMU) exist: East Pacific Ocean and West Pacific Ocean (Wallace et al. 2010; Benson et al. 2011). Although the leatherback is listed as Vulnerable globally in the IUCN Red List, the two Pacific subpopulations are listed as Critically Endangered due to significant declines over the past several decades (Eckert et al. 2012; B P Wallace et al. 2013; Spotila et al. 2000). In the Pacific Ocean, leatherback declines have been estimated at 95% in the last 25 years (Lewison & Crowder 2007; Spotila et al. 2000) while other research has estimated annual longline associated mortality to be between 12 and 27% (Brouwer & Bertram 2009; Kaplan 2005). Thus, fisheries bycatch has been identified as a significant cause of the observed decline and continues to threaten these two subpopulations (Curtis et al. 2015; Wallace et al. 2011; B P Wallace et al. 2013; Donoso & Dutton 2010; Kaplan 2005).

In New Zealand, 288 sighting, stranding and incidental capture (commercial and recreational bycatch) records have been documented from 1892 to 2015 (Gill 1997; Godoy et al. 2011; D. Godoy unpubl. data). This species has been reported from the Kermadec islands (c. 30^{o} S) south to Foveaux Strait (c. 47^{o} S) and east to the Chatham Islands (44^{o} S, 176^{o} W) (Gill 1997; McCann 1966a; Eggleston 1971; Cheeseman 1893; Godoy et al. 2011; D. Godoy, unpubl. data). Despite having a long history of records and a wide distribution in New Zealand, very little local population information exits for this species. However, available data suggests a seasonal influx of adult turtles (μ = 152.1 cm CCL, SD 19.1 cm, range 91.0-195.0 cm, n = 13) which are most often encountered off the North Island during summer and autumn (Gill 1997; D. Godoy, unpubl. data). In addition, preliminary genetic analysis indicates at least some originate from the West Pacific Ocean subpopulation (D. Godoy, unpubl. data). This possible connectivity to west Pacific rookeries is also supported by recent satellite telemetry studies (Benson et al. 2011). Benson et al. (2011) have shown some post-nesting western Pacific females migrate south from their nesting beaches in Papua New Guinea and Solomon Islands into foraging grounds around northern New Zealand. Consequently, available data suggests New Zealand may be an important seasonal foraging ground for adult leatherback turtles.

Results from this study show that leatherback turtles are the most vulnerable species to fisheries bycatch in New Zealand waters, with surface long line activities in FMA 1 (58%, *n* = 52) and FMA 2 (24%, *n* = 22) accounting the vast majority of all leatherback interactions. On average, and excluding 2015/16, 13 turtles were captured each fishing year (SD = 7.2, range = 2-21, *n* = 89). Seasonally, leatherback bycatch was highest during summer and autumn when temperatures were between 22.8 °C (March) and 15.1 °C (June). This period is when foraging adult leatherback turtles have been shown to seasonally migrate south into highly productive temperate waters around New Zealand (Benson et al. 2011). The four incidences of multiple capture events of leatherbacks in FMA 1 and FMA 2 were between February and April, further supporting this clustering may be indicative of the seasonal importance of this region.

Results also show that leatherback bycatch reported by fishers accounted for 93% (*n* = 77) of records during summer and autumn, while only 7% (*n* = 6) records were from CSP observers. Considering this, based on the very low number of CSP Observer reports in comparison to commercial bycatch records during the summer and autumn, the data suggests very low observer coverage during these critical periods when leatherback turtles are most at risk of fisheries interactions in New Zealand waters. In addition, if indeed the seasonal population comprises mostly adult turtles, their loss to the population as a result from bycatch could significantly affect population recovery given that the reproductive values of adult turtles are relatively higher than smaller (younger) turtles (Crouse et al. 1987; Bryan P. Wallace et al. 2013). In conclusion, given the critical population status of Pacific leatherbacks, a lack of information relative to their presence and ecology in New Zealand, and the high number and rate of commercial bycatch in New Zealand, their risk to fisheries bycatch in New Zealand is considered high.

Green turtle (Chelonia mydas)

The green turtle (family: Cheloniidae) is a hard shelled turtle that can be identified by the structure and arrangement of the scutes (scales) of the carapace and the number of prefrontal scales between the eyes (Pritchard & Mortimer 1999). Although other features such as size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics in marine turtles, all should be taken into consideration when identifying this species from other Cheloniids. Green turtles grow to over 1.2 m in total length and to c. 230 kg (Pritchard & Mortimer 1999). Overall generation length for this species have been estimated at c. 35-50 years (Seminoff 2004).

The green turtle has a circum-global distribution, ranging throughout tropical and subtropical waters (Hirth 1997). This species' range can be extensive, occupying coastal nesting areas, oceanic habitats, neritic foraging grounds and migratory pathways throughout their lives (Bolten 2003; Musick & Limpus 1997). Nesting occurs across tropical and subtropical regions between 30° S and 30° N (Hirth 1997). No nesting occurs in New Zealand (Gill 1997; D. Godoy pers. obs.). Upon leaving the nest neonate hatchlings disperse into oceanic developmental habitats for a period of 3-10 years, foraging as epi-pelagic omnivorous macro-planktivores (Boyle & Limpus 2008; Bjorndal 1997; Zug et al. 2002). Eventually, juveniles recruit into nearshore neritic foraging and developmental habitats at approximately 30-45 cm CCL (Musick & Limpus 1997; Balazs 1985; Arthur & Balazs 2008). At this stage they transition into benthic herbivores although some plasticity to their diet has been observed (Cardona et al. 2009; González Carman et al. 2014; Seminoff et al. 2002; Bjorndal 1997). They may remain localised or transition through a series of developmental habitats until they reach maturity at 26-40 years old (Balazs et al. 1987; Limpus & Chaloupka 1997; Koch et al. 2007; Senko et al. 2010; Seminoff et al. 2002). When adults of both sexes reach maturity, they will begin to periodically undertake breeding migrations (at intervals of 2-9 years), often over thousands of kilometres, to their natal rookery (Hirth 1997). They will continue to remigrate between their favoured foraging grounds and their natal rookery for the remainder of their reproductive life.

While the green turtle is a single global species, a clear phylogenetic split exists between the Atlantic Ocean and Pacific Ocean, as well as additional population level genetic differentiation within each ocean basin (Norman et al. 1994; Bowen et al. 1992; Hirth 1997). Seventeen geographically and demographically distinct RMU have been defined, with seven located in the Pacific Ocean region (Wallace et al. 2010). Across the Indo-Pacific Ocean region, approximately 33 genetically distinct breeding stocks have been identified (Jensen 2010; Dethmers et al. 2006; Naro-maciel et al. 2014; Dutton et al. 2014). Individuals from these breeding stocks have been shown to aggregate at mixed stock foraging grounds which span the entire region. To date the largest remaining green turtle rookery in the world is located at Raine Island, Northern Great Barrier reef (C J Limpus 2008a). Despite its widespread distribution and recovery of some subpopulations, the green turtle is listed as globally Endangered in the IUCN Red List due to significant declines over the past several decades (Seminoff 2004). Green turtles are vulnerable to anthropogenic impacts during all life-stages including bycatch in fisheries activities (Seminoff 2004).

In New Zealand, 239 sighting, stranding and incidental capture (commercial and recreational bycatch) records have been documented from 1895 to 2015 (Gill 1997; Godoy et al. n.d.; D. Godoy, unpubl. data). Green turtle records extend from the Kermadec islands (c. 30^{o} S) south to Canterbury (c. 43^{o} S) (Gill 1997; Godoy et al. n.d.). Recent research has described the New Zealand population as a discrete assemblage of post-pelagic immature juveniles to large subadults present year round in its northern waters (c. 34°–38° S) (Godoy et al. n.d.). Unpublished data provides evidence that this population is foraging in nearshore benthic habitats and comprised of mixed stock origins from southwest Pacific and East Pacific rookeries (D. Godoy, unpubl. data).

A total of 12 bycatch records over the eight year period were recorded for this species with an average bycatch of two turtles per fishing year (SD = 0.8, range = 1-3). Although at low levels, bycatch data suggests that both oceanic phase juvenile turtles and post-settlement neritic resident turtles are at risk from fisheries activities in the New Zealand region. In particular, post-settlement juveniles and sub-adults are most likely at risk in northern inshore regions. This is reflected in the cluster of bycatch in the Hauraki Gulf, an area that overlaps with the known distribution of this species in New Zealand. Given the endangered status of this species in the region, limited local population information available, and their low capture rates, their risk to fisheries bycatch in New Zealand is considered moderately low.

Hawksbill turtle (Eretmochelys imbricata)

The hawksbill turtle (family: Cheloniidae) is a hard shelled turtle that can be identified by the structure and arrangement of the scutes (scales) of the carapace and the number of prefrontal scales between the eyes (Pritchard & Mortimer 1999). Although other features such as size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics in marine turtles, all should be taken into consideration when identifying this species from other Cheloniids. Hawksbill turtles grow to about 1 m in total length and typically 60-80 kg (Pritchard & Mortimer 1999). Overall generation length for this species has been conservatively estimated at c. 35-45 years (Mortimer & Donnelly 2008).

The hawksbill turtle has a circum-global distribution throughout tropical and subtropical waters (Mortimer & Donnelly 2008). Nesting occurs across tropical regions mostly scattered on small isolated sandy beaches and in low density (Mortimer & Donnelly 2008; Witzell 1983). No nesting occurs in New Zealand (Gill 1997; D. Godoy pers. obs.). Upon leaving the nest neonate hatchlings disperse into oceanic habitats and complete a juvenile epipelagic stage before recruiting into tidal and subtidal coastal habitats (e.g. coral reefs) at approximately 25-35 cm CCL (Limpus & Fien 2009; Bjorndal 1997). Once hawskbills reach maturity, at approximately 20-40 years, they periodically undertake breeding migrations (at remigration intervals of several years) between foraging areas and their natal nesting rookeries (Witzell 1983; Bowen & Karl 1997). The hawksbill is an omnivorous species feeding on a wide range of sponges, tunicates, molluscs and macroalgae (Bjorndal 1997; Witzell 1983).

While the hawksbill turtle is a single global species, phylogenetic structuring occurs between the Atlantic, Pacific and Indian Ocean basins, as well as at the subpopulation level (Vargas et al. 2015; Duchene et al. 2012; Bowen & Karl 2007). Thirteen geographically and demographically distinct RMU have been defined, with six located in the Pacific Ocean region (Wallace et al. 2010). Within the Indo-Western Pacific Ocean region, Australia contains the largest remaining breeding populations of hawksbill turtles (Limpus & Fien 2009). Due to intense commercial exploitation for tortoiseshell, taxidermied whole animals, habitat destruction, incidental capture in fisheries, and harvest for eggs and meat, this species has experienced significant subpopulation declines across its entire range (Mortimer & Donnelly 2008). Consequently, this species is listed as Critically Endangered in the IUCN Red List (Mortimer & Donnelly 2008). Fisheries bycatch continues to threaten this species, particularly in coastal trawl and gillnet fisheries (Limpus & Fien 2009; Brouwer & Bertram 2009). Information on bycatch in surface longline fisheries of the Western and Central Pacific Ocean is very limited and difficult to quantify due to low observer coverage and a lack of reporting (Limpus & Fien 2009). However, Brouwer and Betram (2009) consider longline bycatch risk to be low for females and juveniles.

In New Zealand, 53 sighting and stranding records have been documented from 1949 to 2015 (Gill 1997; D. Godoy, unpubl. data). No reports of incidental capture in fisheries activities (commercial or recreational) has been documented (Gill 1997; Godoy et al. n.d.). Hawksbill records extend from the Kermadec islands (c. 30^{o} S) south to Palliser Bay, Wellington (c. 41^{o} S) while no records from the South Island have been documented (Gill 1997; Godoy et al. n.d.). Almost no local population information exits for this species in New Zealand. However, available data shows hawksbill distribution is concentrated around Northland with a significant peak in strandings during winter (July-September) (Gill 1997; Godoy et al. n.d.). The observed size structure suggests all turtles are juvenile to large sub-adults (μ = 53.2 cm CCL, SD 14.5 cm, range 35.0-90.0 cm, n = 23). No information exists in terms of ecology, regional connectivity or genetic origin.

A total of six bycatch records over the eight year period were recorded for this species, however there is some uncertainty to the validity of five records (see extent of interactions section). Overall, the risk to hawksbill turtles from fisheries activities in New Zealand is considered low given that a) these species are primarily tropical and low numbers have been recorded in New Zealand, and b) low level bycatch identified in this review. However, given that this species is critically endangered and local population information is very limited, an accurate risk assessment cannot be made.

Loggerhead turtle (Caretta caretta)

The loggerhead turtle (family: Cheloniidae) is a hard shelled turtle that can be identified by the structure and arrangement of the scutes (scales) of the carapace and the number of prefrontal scales between the eyes (Pritchard & Mortimer 1999). Although other features such as size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics in marine turtles, all should be taken into consideration when identifying this species from other Cheloniids. Loggerhead turtles grow to about 1.2 m in total length and to c. 180 kg (Pritchard & Mortimer 1999; Dodd 1988). Overall generation length for this species has been estimated at c. 45 years (Casale & Tucker 2015).

The loggerhead is a single polymorphic species that has a circum-global distribution across tropical, subtropical and temperate waters (Marquez 1990; Dodd 1988; Kobayashi et al. 2014). Nesting occurs across tropical and subtropical regions (C J Limpus 2008b). No nesting occurs in New Zealand (Gill 1997; D. Godoy pers. obs.). Upon leaving the nest neonate hatchlings disperse into oceanic habitats and complete a juvenile epipelagic stage for a highly variable period of 4-19 years (Casale & Tucker 2015). Consequently, loggerheads recruit into neritic foraging and developmental habitats at a size ranging from c. 25 cm CCL in the Mediterranean, 46-64 cm in the western Atlantic, c. 60 cm in Japan, and c. 70 cm in Australia (Conant et al. 2009). Once loggerheads reach maturity, at approximately 10-39 years, they periodically undertake breeding migrations (at remigration intervals of several years) between foraging areas and their natal nesting rookeries (Casale & Tucker 2015; Bowen & Karl 1997; Dodd 1988). The loggerhead is a primarily carnivorous species feeding on a wide range of crustaceans, molluscs, cnidarians, fish and macroalgae (Bjorndal 1997; Dodd 1988; C J Limpus 2008b).

While the loggerhead turtle is a single global species, phylogenetic separation exists between the Atlantic-Mediterranean and Indo-Pacific Ocean basins, as well as at the subpopulation level (Bowen & Karl 2007; Marquez 1990). Ten geographically and demographically distinct RMU have been defined, with two located in the Pacific Ocean region comprising the North Pacific (Japan) and South Pacific (eastern Australia-New Caledonia) breeding stocks (Wallace et al. 2010; C J Limpus 2008b). The loggerhead turtle is listed as Vulnerable globally in the IUCN Red List, however the South Pacific subpopulation is listed as Critically Endangered (Casale & Tucker 2015). Fisheries bycatch was assessed as the most significant threat to loggerhead turtles worldwide (Casale & Tucker 2015; Gilman & Bianchi 2009). In addition, bycatch mortality of oceanic juveniles in the longline fisheries of Chile and Peru is considered a significant threat to population recruitment in the South Pacific subpopulation (Limpus & Casale 2015). Brouwer and Betram (2009) consider shrimp bycatch risk to be high for females and juveniles.

In New Zealand, 55 sighting and stranding records have been documented from 1885 to 2015 (Gill 1997; D. Godoy, unpubl. data). Loggerhead records extend from the Kermadec islands (c. 30^{o} S) south to Stewart Island (c. 47^{o} S) (Gill 1997; Godoy et al. n.d.). Almost no local population information exits for this species in New Zealand. However, available data shows loggerhead distribution is concentrated mainly around the North Island throughout the year (Gill 1997; Godoy et al. n.d.). The observed size structure based on limited samples suggests all turtles are small juveniles to large sub-adults (μ = 40.9 cm CCL, SD 29.1 cm, range 8.0-80.0 cm, n = 16). No information exists in terms of ecology, regional connectivity or genetic origin.

Only two reports of incidental capture in fisheries activities have been documented for this species (this study). This suggests that bycatch risk for this species in New Zealand is low. However, given the significant fisheries risk to loggerhead turtles, the critically endangered status of the South Pacific subpopulation, and the lack of local population information, means an accurate risk assessment cannot be made.

Olive ridley turtle (Lepidochelys olivacea)

The olive ridley (family: Cheloniidae) is a hard shelled turtle that can be identified by the structure and arrangement of the scutes (scales) of the carapace and the number of prefrontal scales between the eyes (Pritchard & Mortimer 1999). Although other features such as size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics in marine turtles, all should be taken into consideration when identifying this species from other Cheloniids. Olive ridley turtles grow to about 0.8 m in total length and typically 35-50 kg (Pritchard & Mortimer 1999). Overall generation length for this species has been estimated at c. 20 years (Seminoff 2004).

The olive ridley turtle has a circum-global distribution in tropical to warm temperate waters (Abreu-Grobois & Plotkin 2008; Bowen et al. 1998). The olive ridley is mainly a carnivorous species feeding on a wide range of fish, salps, crustaceans, molluscs and macroalgae in neritic and epipelagic habitats (Polovina et al. 2004; Colman et al. 2014; Bjorndal 1997; Musick & Limpus 1997). Although found in a range of coastal to oceanic habitats, adults from the eastern Pacific region are predominantly pelagic (Polovina et al. 2004; Plotkin 2010). Nesting occurs across tropical and subtropical regions in *arribada* (mass nesting), dispersed nesting or solitary episodes, and commonly in successive years (Plotkin et al. 1994; Plotkin 2014). No nesting occurs in New Zealand (Gill 1997; D. Godoy pers. obs.).

While the olive ridley is a global species, intra-specific genetic partitioning exists between the Atlantic Ocean and Pacific Ocean, as well as within each ocean basin (Bowen et al. 1998; Shanker et al. 2004). Globally, four main phylogeographic lineages have been identified: Atlantic, east India, Indo-Western Pacific, and eastern Pacific (Bowen et al. 1998; Shanker et al. 2004; Hahn et al. 2012). Eight geographically and demographically distinct RMU have been defined, with two located in the Pacific Ocean region comprising the West Pacific and East Pacific breeding stocks (Wallace et al. 2010). In the southwest Pacific, the main nesting rookeries occur in Northern Australia and Indonesia (Colin J Limpus 2008; Plotkin 2014). The olive ridely is considered the most abundant of all marine turtle species, yet quantitative validation of global population estimates is complex and may overemphasise the contribution of some populations while under representing others (Abreu-Grobois & Plotkin 2008). Despite its estimated abundance the overall population trend is in decline and therefore listed as globally Vulnerable in the IUCN Red List (Abreu-Grobois & Plotkin 2008). Fisheries impacts through bycatch and entanglement has contributed to the observed decline, and continues to threaten many subpopulations (e.g. Orissa, India) (Abreau-Grobois & Plotkin 2008; Plotkin 2014).

In New Zealand, 29 sighting and stranding records have been documented from 1956 to 2015 (Gill 1997; D. Godoy, unpubl. data). No reports of incidental capture in fisheries activities (commercial or recreational) has been documented (Gill 1997; Godoy et al. n.d.). Olive ridley records extend from Northland (c. 35^{o} S) south to Stewart Island (c. 47^{o} S) and east to the Chatham Islands (44^{o} S, 176^{o} W) (Gill 1997; Godoy et al. n.d.). Almost no local population information exits for this species in New Zealand. Limited available data from stranded turtles show they most often strand during winter (July-September) and have all been sub-adult or mature adults (μ = 64.7 cm CCL, SD = 1.7, range = 52.5–85.0 cm, n = 16) (D.

Godoy, unpubl. data). No information exists in terms of ecology, regional connectivity or genetic origin.

No commercial bycatch was reported for this species over the 8 year period of this study. Given the vulnerable status of this species in the region, very limited population information available for olive ridley turtles in New Zealand, and the lack of any bycatch reported, their risk to fisheries bycatch in New Zealand is considered low.

Recommendations

This research has identified that the surface longline fishery targeting swordfish and tunas poses the greatest risk of bycatch for marine turtles in New Zealand. In addition, bycatch risk is higher in the oceanic habitats of FMA 1 and FMA 2 during summer and autumn. In particular, the critically endangered Pacific leatherback turtle is incidentally captured most often and therefore most at risk. Results also show that observer coverage does not adequately overlap the fishery, FMA, or season where most bycatch occurs. To address these issues in order to reduce the overall bycatch of marine turtles in New Zealand waters several recommendations are made.

1. Implement and monitor a minimal marine turtle interaction rate

The Western and Central Pacific Fisheries Commission (WCPFC) recommend commission members (including New Zealand) to implement measures to reduce marine turtle bycatch in shallow-set longline fisheries (Brouwer & Bertram 2009). Accordingly, under resolution RES2005-04 (Western and Central Pacific Fisheries Commission 2005) and conservation and management measure CMM2008-03 (CMM2008-03 2008) the WCPFC tasks the Scientific Committee to recommend a “minimal” (maximum acceptable rate) marine turtle interaction rate for shallow-set longline fisheries (Brouwer & Bertram 2009). Thus, the commission proposes an interaction rate of 0.019 turtles (all species combined) per 1000 hooks or less for shallow-set longline fisheries targeting swordfish in the Western and Central Pacific Ocean (WCPO).

It is evident that the interaction rate of marine turtles in the New Zealand SLL fishery exceeds the proposed annual minimal marine turtle interaction rate in certain FMA. Therefore, it is recommended that an interaction target rate of 0.019 turtles or less is achieved. The target rate should be calculated per FMA rather than the fishery as a whole in order to account for the heterogeneous distribution of marine turtles across different FMA.

If the interaction rate exceeds the recommended minimal limit as prescribed in the CMM2008-03, then appropriate mitigation actions should be considered. For example, the data suggests that area/time closures in FMA where interaction rates exceed prescribed limits may be suitably tailored in the New Zealand context. Similar management actions have shown to significantly reduce marine turtle bycatch in the Hawaiian shallow-set longline fishery and the U.S. west coast (Curtis et al. 2015; Gilman et al. 2007).

2. Implement the Guidelines to Reduce Sea Turtle Mortality

As part of conservation and management measure CMM2008-03 (CMM2008-03 2008), WCPFC commission members are to adopt the United Nations Food and Agriculture Organisation (FAO) *Guidelines to Reduce Sea Turtle Mortality* where appropriate. Given that SLL activities in New Zealand have resulted in higher than recommended interaction rates, it is recommended that mitigation measures outlined in the guidelines are investigated. For example, key measures for surface longline activities include:

a. Investigate the use of wide circle hooks instead of J hooks or tuna hooks. Evidence suggests incidental capture rates of marine turtles is significantly reduced without compromising target catch rates (Gilman et al. 2007; Read 2007; Anon 2006). Using large circle hooks (e.g. 18/0) has also been shown to significantly reduce hook ingestion (leading to increased mortality) and the entanglement of marine turtles, particularly leatherbacks (Gilman 2011; Read 2007). Bycatch of other protected species (e.g. sharks) may also benefit from using wide circle hooks (Gilman et al. 2007).

b. Investigate the use of fish bait instead of squid bait. Research suggests incidental capture rates of marine turtles can be significantly reduced when squid bait is replaced with fish bait (Gilman et al. 2007). In addition, when fish bait is used in conjunction with wide circle hooks, capture rates can be further reduced (Gilman et al. 2007).

3. Review the allocation of observer coverage

Observer coverage in the domestic longline fleet is very low yet accounts for the highest number of marine turtle bycatch. In addition, marine turtle bycatch is highest in FMA 1 and FMA 2 during summer and autumn. Therefore it is recommended that observer coverage is allocated more appropriately in order to achieve greater proportional coverage in these areas during high risk periods. Thus, more robust data will be collected and validated by trained observers.

4. Improve data quality and reporting

Improving data quality and reporting will provide a more accurate assessment of protected species bycatch. In relation to marine reptile bycatch in New Zealand fisheries some areas that can be improved include:

a. Species identification: it is highly unlikely that leatherback turtles were misidentified either by observers or fishers given their size and unique morphological characteristics in comparison to other marine turtle species. In contrast, however, the misidentification of cheloniid species is highly likely given their morphological similarities. Therefore, the utility of the data collected for assessing species or population specific impacts will be limited if species identification cannot be validated. To reduce the likelihood of species misidentification, ensuring observers are adequately trained and appropriate information (e.g. identification guides) are provided to fishers is critical. In addition, photographs of incidentally captured species should be taken wherever possible to validate species identification.

b. Biological data: where possible, biometric measurements and tissue samples for genetic analysis should be collected. If dead animals are landed onboard, it is recommended they are made available to researchers for necropsy.

c. Bycatch report forms: the information provided by observers and fishers on their respective bycatch forms were contradictory in some instances due to obscure or misleading field codes. For example, in some cases, hooked turtles were reported as sustaining injuries yet were also reported as being released alive and uninjured. In other instances, turtles that were released with the hook and snood still were reported as having being released alive and unharmed. Overall, 88% of all by-caught marine turtles were reported as uninjured, and with no additional information as to the capture type or release method used. This may misrepresent the true extent of bycatch impacts given that post-release mortality is unknown. Therefore, it is advisable to list any hooked animals as released alive and injured, with a coded description of the injury.

5. *Improve population information and research*

Given the lack of population information available for all marine reptile species present in New Zealand waters, it is recommended to undertake research to enable more accurate fisheries risk assessments to be made. This could include research to understand population structure, spatio-temporal distribution and regional connectivity. Because evidence suggests New Zealand is an important seasonal foraging ground for critically endangered western Pacific leatherback turtles and given the high interaction rate of bycatch in local fisheries, research on this species should be considered a priority.

References

Abreu-Grobois A., Plotkin P T, 2008. *Lepidochelys olivacea*: The IUCN Red List of Threatened Species 2008. IUCN SSC Marine Turtle Specialist Group, pp.6–8.

Anon, 2006. The sea turtle bycatch mitigation program for the coastal longline fleets and preliminary results of circle hook experiments. Inter-American Tropical Tuna Comission: Working Group on Bycatch 5th meeting, Busan, Korea.

Arthur K E, Balazs G H, 2008. A comparison of immature green turtle (*Chelonia mydas*) diets among seven sites in the main Hawaiian Islands. Pacific Science, 62(2), pp.205–217.

Avens L, Taylor J. C, Goshe L R., Jones T. T, Hastings M, 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research, 8(3), pp.165–177.

Balazs G H, Forsyth RG, Kam A K H, 1987. Preliminary assessment of habitat utilization by Hawaiian green turtles in their resident foraging pastures, Honolulu. NOAA Technical Memorandum: TM-NMFS-SWFC-71 .

Balazs G H, 1985. Status and ecology of marine turtles at Johnston Atoll. Atoll Research Bulletin 285, p. 49.

Benson S R, Dutton P H, Hitipeuw C et al., 2007. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology, 6(1), pp.150–154.

Benson S R, Eguchi T, Foley D G et al., 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere, 2(7), pp.1–27.

Bjorndal K A, 1997. Foraging ecology and nutrition of sea turtles. In P. L. Lutz & J. A. Musick, eds. *The biology of sea turtles.* Boca Raton, Florida.: CRC Press, pp. 199–231.

Block B A, Jonsen I D, Jorgensen S J et al., 2011. Tracking apex marine predator movements in a dynamic ocean. Nature, 475(7354), pp.86–90.

Bolten A B, 2003. Variation in sea turtle life history patterns: neritic vs. oceanic developmental stages. In P. L. Lutz, J. A. Musick, & J. Wyneken, eds. *The Biology of Sea Turtles, Volume II*. Boca Raton, Florida.: CRC Press., pp. 243–257.

Bowen B W, Clark a M, Abreu-Grobois F a et al., 1998. Global phylogeography of the ridley sea turtles (*Lepidochelys* spp.) as inferred from mitochondrial DNA sequences. Genetica, 101(3), pp.179–189.

Bowen B W, Karl S A, 1997. Population genetics, phylogeography, and molecular evolution. In P. L. Lutz & J. A. Musick, eds. *The biology of sea turtles*. Boca Raton, Florida: CRC Press, p. 432.

Bowen B W, Karl S A, 2007. Population genetics and phylogeography of sea turtles. Molecular Ecology, 16(23), pp.4886–4907.

Bowen B W, Meylan A B, Ross J P et al., 1992. Global population structure and natural history of the green turtle (*Chelonia mydas*) in terms of matriarchal phylogeny. Evolution, 46(4), pp.865–881.

Boyle M C, Limpus C J, 2008. The stomach contents of post-hatchling green and loggerhead sea turtles in the southwest Pacific: an insight into habitat association. Marine Biology, 155(2), pp.233–241.

Brouwer S, Griggs L, 2009. Description of New Zealand’s shallow-set longline fisheries. In *WCPFC Scientific Committee Fifth Regular Session*. p. 43.

Brouwer S, Bertram I, 2009. Setting bycatch limits for sea turtle in the western and central pacific oceans shallow-set longline fisheries. In *WCPFC Scientific Committee Fifth Regular Session*. p. 11.

Cardona L, Aguilar A, Pazos L, 2009. Delayed ontogenic dietary shift and high levels of omnivory in green turtles (*Chelonia mydas*) from the NW coast of Africa. Marine biology 156, (7). PP. 1487–1495.

Carriol R P, Vader W, 2002. Occurrence of Stomatolepas elegans (Cirripedia: Balanomorpha) on a leatherback turtle from Finnmark, northern Norway. Journal of the Marine Biological Association of the United Kingdom, 82(6), pp.1033–1034.

Casale Paolo, Tucker Anton D., 2015. *Caretta caretta*. The IUCN Red List of Threatened Species 2015. The IUCN Red List of Threatened Species 2015, 8235, p.20.

Cheeseman T F, 1893. On the occurrence of the luth, or leathery turtle, on the coasts of New Zealand. Transactions of the New Zealand Institute, 25, pp.108–110.

CMM2008-03, 2008. Conservation and Management Measure of Sea Turtles. In *WCPFC Fifth Regular Session*. pp. 1–5.

Cogger H G, Heatwole H F, 2006. *Laticauda frontalis* (de Vis, 1905) and *Laticauda saintgironsi n*.sp. from Vanuatu and New Caledonia (Serpentes: Elapidae: *Laticaudinae*) — a New Lineage of Sea Kraits? Records of the Australian Museum, 58, pp.245–256.

Cogger H, 2007. Marine snakes. In V. Tzioumis & S. Keable, eds. *Description of Key Species Groups in the East Marine Region*. p. 289.

Colman Liliana Poggio, Sampaio Cláudio Luis S, Weber Marilda Inês, de Castilhos Jaqueline Comin, 2014. Diet of Olive Ridley Sea Turtles, *Lepidochelys olivacea*, in the Waters of Sergipe, Brazil. Chelonian Conservation and Biology, 13(2), pp.266–271.

Conant T A., Dutton P H., Eguchi T et al., 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Biological Review Team to the National Marine Fisheries Service. p. 222 pp.

Crouse D T, Crowder L B, Caswell H, 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology, 68(5), pp.1412–1423.

Curtis K Alexandra, Moore Jeffrey E, Benson S R, 2015. Estimating Limit Reference Points for Western Pacific Leatherback Turtles (*Dermochelys coriacea*) in the U.S. West Coast EEZ. PLoS ONE, 10(9).

Davenport J, 1998. Sustaining endothermy on a diet of cold jelly: Energetics of the leatherback turtle *Dermochelys coriacea*. Herpetological Bulletin, (62), pp.4–8.

Davenport J, 1997. Temperature and the life-history strategies of sea turtles. Journal of Thermal Biology, 22(6), pp.479–488.

Dethmers K E M, Broderick D, Moritz C et al., 2006. The genetic structure of Australasian green turtles (*Chelonia mydas*): Exploring the geographical scale of genetic exchange. Molecular Ecology, 15(13), pp.3931–3946.

Dodd K C., 1988. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). US Fish and Wildlife Service, 88(14), p.110.

Donoso M, Dutton P H, 2010. Sea turtle bycatch in the Chilean pelagic longline fishery in the southeastern Pacific: Opportunities for conservation. Biological Conservation, 143(11), pp.2672–2684.

Duchene S, Frey A, Alfaro-Núñez A et al., 2012. Marine turtle mitogenome phylogenetics and evolution. Molecular Phylogenetics and Evolution, 65(1), pp.241–250.

Dunson W A, 1975. The Biology of Sea Snakes. University Park Press, Baltimore.

Dutton P H, Bowen B W, Owens D W, Barragan A, Davis S K, 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology, 248(3), pp.397–409.

Dutton P H, Jensen M P, Frutchey K et al., 2014. Genetic Stock Structure of Green Turtle (*Chelonia mydas*) Nesting Populations Across the Pacific Islands. Pacific Science, 68(4), pp.451–464.

Eckert K L, 1995. Anthropogenic Threats to Sea Turtles. In K. A. Bjorndal, ed. *Biology and conservation of sea turtles*. Smithsonian Institution Press, Washington D.C. pp. 611–612.

Eckert K L, Wallace B P, Frazier J G, Eckert S A, Pritchard P C H, 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). Biological Technical Publication, (BTP-R4015-2012), p. 158.

Eggleston D, 1971. Leathery Turtle (Reptilia: *Chelonia*) in Foveaux Strait. New Zealand Journal of Marine and Freshwater Research, 5(3 & 4), pp.522–523.

Gaspar P, Benson S R, Dutton P H et al., 2012. Oceanic dispersal of juvenile leatherback turtles: Going beyond passive drift modeling. Marine Ecology Progress Series, 457, pp.265–284.

Gill B J, 1997. Records of turtles and sea snakes in New Zealand, 1837-1996. New Zealand Journal of Marine and Freshwater Research, 31(4), pp.477–486.

Gilman E, Bianchi G, 2009. Guidelines to reduce sea turtle mortality in fishing operations. FAO Fisheries Report, Rome. p. 128.

Gilman E L, 2011. Bycatch governance and best practice mitigation technology in global tuna fisheries. Marine Policy, 35(5), pp.590–609.

Gilman E, Clarke S, Brothers N et al., 2008. Shark interactions in pelagic longline fisheries. Marine Policy, 32(1), pp.1–18.

Gilman E, Kobayashi D, Swenarton T et al., 2007. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation, 139(1–2), pp.19–28.

Godoy D, Richards R, Mackay K T, 2011. Leatherback turtles (*Dermochelys coriacea*) at the Chatham Islands, New Zealand. 2011 SWOT Grant: Final Report,

Godoy D, Smith A, Limpus C J, Stockin K, 2016. The spatio-temporal distribution and population structure of green turtles (*Chelonia mydas*) in New Zealand. New Zealand Journal of Marine and Freshwater Research, 50 (4).

González Carman V, Botto F, Gaitán E et al., 2014. A jellyfish diet for the herbivorous green turtle *Chelonia mydas* in the temperate SW Atlantic. Marine Biology, 161(2), pp.339–349.

Guinea M, Lukoschek V, Cogger H et al., 2010. *Pelamis platura*. The IUCN Red List of Threatened Species 2010: e.T176738A7293840.,

Hahn A T, Jensen M, Bowen B W et al., 2012. Preliminary results on global genetic structure and phylogeography of olive ridleys (*Lepidochelys olivacea*). In T. T. Jones & B. P. Wallace, eds. *Proceedings of the Thirty-First Annual Symposium on Sea Turtle Biology and Conservation*. NOAA Technical Memorandum NMFS-SEFSC-631, p. 207.

Harley S J, Kendrik T, 2006. Characterization of sea turtle bycatch in New Zealand's tuna fisheries. WCPFC Scientific Committee Second Regular Session, Manila, Philippines, WCPFC-SC2-2006/EB WP-3, p.7.

Hirth H F, 1997. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). Biological Report 97(1). U.S. Fish and Wildlife Service, Washington, D.C.

Hitchmough R A, Anderson P J, Barr B et al., 2013. Conservation status of New Zealand reptiles, 2012.Department of Conservation, Wellington.

Hochscheid S, Bentivegna F, Speakman J R, 2002. Regional blood flow in sea turtles: Implications for heat exchange in an aquatic ectotherm. Physiological and Biochemical Zoology, 75(1), pp.66–76.

James M C, Ottensmeyer C A, Myers R A, 2005. Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. Ecology Letters, 8(2), pp.195–201.

Jensen M P, 2010. Assessing the composition of green turtle (*Chelonia mydas*) foraging grounds in Australasia using mixed stock analyses. University of Canberra, Australia.

Kaplan I C, 2005. A risk assessment for Pacific leatherback turtles (*Dermochelys coriacea*). Canadian Journal of Fisheries and Aquatic Sciences, 62(8), pp.1710–1719.

Kobayashi D R, Farman R, Polovina J J. et al., 2014. "Going with the flow" or not: Evidence of positive rheotaxis in oceanic juvenile loggerhead turtles (*Caretta caretta*) in the South Pacific Ocean using satellite tags and ocean circulation data. PLoS ONE, 9(8).

Koch V, Brooks L B, Nichols W J, 2007. Population ecology of the green/black turtle (*Chelonia mydas*) in Bahía Magdalena, Mexico. Marine Biology, 153(1), pp.35–46.

Lewison R L., Crowder L B., 2007. Putting longline bycatch of sea turtles into perspective.

Conservation Biology, 21(1), pp.79–86.

Limpus C J, Casale P, 2015. *Caretta caretta* (South Pacific subpopulation). The IUCN Red List of Threatened Species 2015: e.T84156809A84156890.,

Limpus C J, Chaloupka M Y, 1997. Nonparametric regression modelling of green sea turtle growth rates (southern Great Barrier Reef). Marine Ecology Progress Series, 149(1–3), pp.23–34.

Limpus C J, 2008a. A biological review of Australian marine turtle species. 2. Green turtle, *Chelonia mydas* (Linnaeus). Queensland Environmental Protection Agency, Brisbane. p.95

Limpus C J, 2008b. A biological review of Australian marine turtles. 1. Loggerhead turtle, *Caretta caretta* (Linnaeus), Queensland Environmental Protection Agency, Brisbane. p.66.

Limpus C J., Fien L, 2009. A Biological Review of Australian Marine Turtles: Hawksbill turtle. Queensland Environmental Protection Agency, Brisbane. p.53.

Limpus C J, 2008. A biological review of Australian marine turtles. 4: Olive Ridley turtle, *Lepidochelys olivacea*. Queensland Environmental Protection Agency, Brisbane. p.27.

Marquez R M, 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. FAO, Rome. p. 81.

McCann C, 1966a. Key to the marine turtles and snakes occurring in New Zealand. Tuatara, 14(2), p.1966.

McCann C, 1966b. The marine turtles and snakes occurring in New Zealand. Tuatara, 5(21), pp.201–215.

Milton D A, 2001. Assessing the susceptibility to fishing of populations of rare trawl bycatch: sea snakes caught by Australia’s Northern Prawn Fishery. Biological Conservation, 101(3), pp.281–290.

Mortimer J A., Donnelly M, 2008. Eretmochelys imbricata. The IUCN Red List of

Threatened Species 2008: e.T8005A12881238. The IUCN Red List of Threatened Species, 8235.

Mrosovsky N, 1980. Thermal Biology of Sea Turtles. American Zoologist, 20(3), pp.531–547.

Musick J A, Limpus C J, 1997. Habitat utilization and migration in juvenile sea turtles P. L. Lutz & J. A. Musick, eds., CRC Press, Boca Raton, FL.

Naro-maciel E, Gaughran S J, Putman N F et al., 2014. Predicting connectivity of green turtles at Palmyra Atoll, central Pacific: a focus on mtDNA and dispersal modelling. Journal of the Royal Society, Interface, 11.

Norman J A, Moritz C, Limpus C J, 1994. Mitochondrial DNA control region polymorphisms: Genetic markers for ecological studies of marine turtles. Molecular Ecology, 3(4), pp.363–373.

Plotkin P T., 2010. Nomadic behaviour of the highly migratory olive ridley sea turtle *Lepidochelys olivacea* in the eastern tropical Pacific Ocean. Endangered Species Research, 13(1), pp.33–40.

Plotkin P T, Byles R A, Owens D W, 1994. Migratory and reproductive behavior of *Lepidochelys olivacea* in the eastern Pacific Ocean. In B. A. Schroeder & B. E. Witherington, eds. *Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation*. NOAA Technical Memorandum NMFS-SEFSC-341, p. 138.

Plotkin P T, 2014. Olive Ridley Sea Turtle (*Lepidochelys olivacea*). 5-year review: summary and evaluation, Maryland and Florida.

Polidoro B A, Elfes C T, Sanciangco J C, Pippard H, Carpenter K E, 2011. Conservation Status of Marine Biodiversity in Oceania: An Analysis of Marine Species on the IUCN Red List of Threatened Species. Journal of Marine Biology, 2011, pp.1–14.

Polovina J J, Balazs G H, Howell E A et al., 2004. Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. Fisheries Oceanography, 13(1), pp.36–51.

Pritchard P C H, 1997. Evolution, phylogeny, and current status. In P. L. Lutz & J. A. Musick, eds. *The biology of sea turtles*. CRC Press, Boca Raton, Florida. p. 432.

Pritchard Peter C H, Mortimer Jeanne A, 1999. Taxonomy, external morphology, and species identification. In K. L. Eckert et al., eds. *Research and Managment Techniques for the Conservation of Sea Turtles*. IUCN/SSC Marine Turtle Specialist Group.

Read Andrew J., 2007. Do circle hooks reduce the mortality of sea turtles in pelagic longlines? A review of recent experiments. Biological Conservation, 135(2), pp.155–169.

Saba V S, Shillinger G L, Swithenbank A M et al., 2008. An oceanographic context for the foraging ecology of eastern Pacific leatherback turtles: Consequences of ENSO. Deep-Sea Research Part I: Oceanographic Research Papers, 55(5), pp.646–660.

Seminoff J A, Resendiz A, Nichols W J, 2002. Diet of east Pacific green turtles (*Chelonia mydas*) in the central Gulf of California, México. Journal of Herpetology, 36(3), pp.447–453.

Seminoff J A, 2004. *Chelonia mydas*. The IUCN Red List of Threatened Species 2004: e.T4615A11037468.,

Senko J, López-Castro M C, Koch V, Nichols W J, 2010. Immature east Pacific green turtles (*Chelonia mydas*) use multiple foraging areas off the Pacific Coast of Baja California Sur, Mexico: First evidence from mark-recapture data. Pacific Science, 64(1), pp.125–130.

Shanker K., Ramadevi J., Choudhury B. C., Singh L., Aggarwal R. K., 2004. Phylogeography of olive ridley turtles (*Lepidochelys olivacea*) on the east coast of India: Implications for conservation theory. Molecular Ecology, 13(7), pp.1899–1909.

Spotila J R, Reina R D, Steyermark A C, Plotkin P T, Paladino F V, 2000. Pacific leatherback turtles face extinction. Nature, 405(6786), pp.529–30.

Vargas Sarah M, Michael P, Ho Simon Y W et al., 2015. Phylogeography, Genetic Diversity, and Management Units of Hawksbill Turtles in the Indo-Pacific. Journal of Heredity, pp.1–15.

Wallace B P, DiMatteo A D, Bolten A B et al., 2011. Global Conservation Priorities for Marine Turtles. PLoS ONE, 6(9), p.e24510.

Wallace B P, DiMatteo A D, Hurley B J et al., 2010. Regional Management Units for Marine Turtles: A Novel Framework for Prioritizing Conservation and Research across Multiple Scales. PLoS ONE, 5(12), pp.1–11.

Wallace B P, Tiwari M, Girondot M, 2013. *Dermochelys coriacea*, The IUCN Red List of Threatened Species 2013: e.T6494A43526147.,

Wallace B P., Kot C Y., DiMatteo A D. et al., 2013. Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. Ecosphere, 4(3), p.art40.

Western and Central Pacific Fisheries Commission, 2005. Resolution 2005-04: Resolution to Mitigate the Impact of Fishing for Highly Migratory Fish Species on Sea Turtles. In *WCPFC Second Regular Session*. pp. 1–3.

Witzell W.N, 1983. Synopsis of biological data on the hawksbill turtle *Eretmochelys imbricata* (Linnaeus, 1766). , 5(31), p.86.

Zug G R, Balazs G H, Wetherall J A, Parker D M, Murakawa S K K, 2002. Age and growth of Hawaiian green seaturtles (*Chelonia mydas*): An analysis based on skeletochronology. Fishery Bulletin, 100(1), pp.117–127.

Appendix

Combined annual surface longline fishing effort (hooks set) for domestic and foreign charter by FMA, observed effort, and marine reptile captures and capture rates (reptiles.1000 hooks set) from 1 July 2008 to 30 November 2015 (*n* = 109). Data includes all commercial and observer records except data where FMA was not reported (i.e. 14,050 hooks).

Fishing year	FMA	Total hooks set	Observed hooks	% observed	Total	Total Catch rate
2008/09	1	13,690	1,000	7.3	0	0.0000
	2	4,790	1,000	20.9	0	0.0000
	7	4,080	0	0.0	0	0.0000
	8	1,650	0	0.0	0	0.0000
	9	12,700	2,050	16.1	0	0.0000
	10	5,400	0	0.0	0	0.0000
2009/10	**1**	26,930	950	**3.5**	1	**0.0371**
	2	45,870	0	0.0	0	0.0000
	7	11,650	0	0.0	0	0.0000
	8	1,100	0	0.0	0	0.0000
	9	45,080	1,240	2.8	0	0.0000
	10	10,000	0	0.0	0	0.0000
2010/11	**1**	45,420	0	**0.0**	2	**0.0440**
	2	15,050	700	4.7	0	0.0000
	7	29,400	5,100	17.3	0	0.0000
	9	48,230	9,550	19.8	0	0.0000
	10	18,850	0	0.0	0	0.0000
2011/12	**1**	31,518	3,988	**12.7**	2	**0.0635**
	2	15,400	0	0.0	0	0.0000
	7	51,470	25,900	50.3	0	0.0000
	8	4,000	4,300	107.5	0	0.0000
	9	40,210	0	**0.0**	2	**0.0497**
	10	17,600	0	0.0	0	0.0000
2012/13	**1**	87,860	6,750	**7.7**	7	**0.0797**
	2	47,630	0	**0.0**	3	**0.0630**
	7	85,270	0	0.0	1	0.0117
	8	11,780	0	**0.0**	1	**0.0849**
	9	116,460	9,850	8.5	2	0.0172
	10	5,900	0	0.0	0	0.0000
2013/14	1	46,328	5,550	12.0	0	0.0000
	2	31,150	0	0.0	0	0.0000
	7	72,870	3,300	4.5	0	0.0000
	8	3,025	0	0.0	0	0.0000
	9	40,180	500	**1.2**	2	**0.0498**
	10	8,200	0	0.0	0	0.0000
2014/15	**1**	156,592	8,850	**5.7**	6	**0.0383**
	2	35,100	0	0.0	0	0.0000
	7	124,600	11,200	9.0	0	0.0000
	8	3,000	0	0.0	0	0.0000
	9	127,600	0	0.0	0	0.0000
2015/16[¥]	1	900	0	0.0	0	0.0000
	9	8,770	0	0.0	0	0.0000
Total		**1,470,993**	**97,728**	**6.6**	**29**	**0.0197**

¥ Fishing year 2015/16 only covers 5 months (1 July 2015 – 30 November 2015).