The ecology and conservation of green turtles (*Chelonia mydas*) in New Zealand

A thesis submitted in partial fulfilment of the requirements

for the degree of

Doctor of Philosophy in Marine Ecology

at Massey University, Albany, New Zealand

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2016

Frontispiece

In the southwestern Pacific Ocean at the edge of their range, post-pelagic green turtles (*Chelonia mydas*) recruit into neritic developmental habitats of northern New Zealand. Photo cover of New Zealand Journal of Marine and Freshwater Research 2016, 50(4), 549-565, courtesy of Steve Hathaway.

Abstract

Historically, little consideration has been given to the occurrence, ecology or conservation of the green turtle (*Chelonia mydas*) in temperate New Zealand. Located geographically at the southern boundary of the distributional range of green turtles in the southwestern Pacific, reports of this species in New Zealand are often overlooked as occasional visitors or stragglers incidentally carried by ocean currents. This convention may be reasonable when considering the temperature constrained distribution of this poikilothermic marine reptile. Despite this, green turtles have been reported in New Zealand waters for more than 100 years, yet no study has undertaken any in depth investigation as to their occurrence in this region. Therefore, this thesis investigated the presence of green turtles in New Zealand waters to test the hypothesis that their occurrence is ephemeral and incidental.

Opportunistic data and samples collected between 1895 and 2013 was collated, reviewed and analysed to investigate several lines of empirical enquiry, including spatio-temporal distribution, population structure, genetic origin, diet composition and anthropogenic effects. Sighting, stranding, and incidental capture revealed a year round presence of post-pelagic immature juveniles to large sub-adult green turtles across northern New Zealand (ca. 34°-38° S). Such occurrence exists despite sea surface temperatures averaging only 14 °C during austral winters. The aggregation exhibited a female:male sex ratio of 1.7:1 which is similar to that reported from proximate warm temperate foraging grounds in eastern Australia. Size frequency data indicated that green turtles recruit to neritic habitats of the North Island at ca. 40.8 cm curved carapace length. This reflects a natural post-oceanic settlement pattern rather than oceanic-phase stragglers incidentally blown ashore by storm and other stochastic events. Supporting this rationale for natural recruitment, diet component data demonstrates that once green turtles settle into New Zealand's nearshore coastal habitats, they transition to a benthic foraging strategy. Notably, green turtles in New Zealand do not ontogenetically transition from omnivory to obligate herbivory with age, but instead consume a variable diet of primarily macroalgae and benthic macro invertebrates. Overall, the confirmation of feeding in New Zealand substantially extends the southern foraging limit for green turtles in the Pacific Ocean.

Genetic analyses of ~770 base pair sequences of mitochondrial (mt) DNA was conducted on 42 stranded green turtles to characterize the genetic structure of this aggregation. Results identified 15 haplotypes including one orphan haplotype from widely dispersed green turtle stocks across the western, central, and eastern Pacific Ocean. When compared to other regional nesting rookeries and foraging grounds, the New Zealand aggregation exemplified its unique composition, predominantly due to the large proportion of haplotypes from the endemic eastern Pacific clade. These results provide a genetic link to east Pacific stocks in the southwestern Pacific; identifying previously undefined regional connectivity and trans-oceanic dispersal for eastern Pacific green turtles.

In order to assess potential human impacts, gross necropsies were conducted on green turtles found stranded in northern New Zealand between 2007 and 2013. Anthropogenic effects predominantly associated with the ingestion of plastic marine debris were identified as the likely cause for the majority of strandings in the North Island. Propeller strike and incidental capture in recreational fisheries were further shown to impact green turtles, particularly for turtles inhabiting neritic habitats adjacent to densely populated urban centres of northeastern New Zealand.

Overall, data presented here supports the hypothesis that New Zealand northern neritic habitats provide a transitional developmental habitat for immature green turtles at the edge of their range in the southwestern Pacific Ocean. Genetic analysis reveals this aggregation is unique when compared to other regional foraging grounds, exhibiting links to discrete genetic stocks from across the Pacific Ocean. In addition, the cause of the exponential increase in records observed over time remains unclear, therefore warrants further research and monitoring of this endangered marine reptile; particularly in light of climate-mediated environmental change presently experienced in the region.

Acknowledgements

"If you change, the world changes."

It's hard to know when my journey began, maybe it was when I was young when I watched the undersea adventures of Jacque Cousteau or David Attenborough's *'The Living Planet'*; maybe it evolved later, during the long night watches aboard the *R Tucker Thompson* as we sailed to protest French nuclear testing in Moruroa. Whatever the catalyst, it has led me to this point, a culmination of many years work and stubborn determination to make a difference. Getting this far was no small feat, and certainly would not have been achievable without the support, inspiration and belief of many people along the way. At its inception, the research would have never got off the ground were it not for Craig Thorburn and Mike Bhana who dusted off my satellite tracking proposal and made it part of *'Ocean Zoo'*. Like me, you both saw its research and awareness potential; and let's face it we all love a good adventure. Thanks also to Andrew Baker and Andrew Christie (AJ) for the support and laughs along the way. A big thanks to Kelly Tarlton's and its staff for all the support (and laughs) and rehabilitation of marine turtles over the years – how things have changed for the better! As the project gained momentum, thanks to Steve Cook (Cookie) and Steve O'Shea from Auckland University of Technology (AUT) for pushing me to take on the PhD; looking back now I can thank you although a few years ago I wasn't so sure.

To Dr. Karen Stockin, my sincerest gratitude for not only being my primary supervisor, but also for being a great mentor and continually believing in me and the project. Also a special thanks to you for having my back when things did not progress as planned. A big thanks to Prof Dianne Brunton, my co-supervisor, who, after a casual catch up in 2010, invited me to rekindle the research at Massey. Without your initial support, this thesis simply would not have eventuated. There have also been a number of people and organisations that have supported the research over the years including staff from the New Zealand Centre for Conservation Medicine at Auckland Zoo, the Department of Conservation (DOC) particularly staff from the Marine Conservation Unit, permitting, Auckland Conservancy/ field office and the Kaitaia field office. A special thanks to Clinton Duffy for the official and unofficial support over the years – now I'll hopefully have time to dedicate to the *Bigfish* project. A sincere thanks to all the Iwi across the country for their ongoing support for the research and conservation of marine turtles in New Zealand, particularly to the tangata whenua of Te Tai Tokerau (Northland): Ngāti Kuri, Te Aupōuri, Ngai Takoto, Te Rarawa, Te Roroa, Te Uri-o-Hau, Ngāti Wai; and Tamaki Makaurau: Ngāti Whatua, Ngāti Manuhiri.

My thanks also extend to the wider community of the North Island who have been pivotal in the success of the project through their ongoing support, interest and actively reporting sightings of marine turtles and rescuing stranded animals. Such support would not have gained such traction if it were not for the TVNZ, Wild Film, TV and print media journalists and filmmakers who provided a platform to raise awareness and highlight the plight of marine turtles and the ocean. I also thank the many school kids around the country that I have been lucky enough to share stories with and inspire, because in turn, they have been a major source of inspiration to me, their honesty and appreciation is humbling. A massive thanks also go to those who fighting the good fight, the likes of Hayden and Ben from Sea Cleaners; Sam and Camden from Sustainable Coastlines, Samara and Kim from Experiencing Marine Reserves (EMR)/ Whitebait Connection; Schannel, Sue and Olive from Conservation International, and the huge number of teachers who inspire their students.

A project of this magnitude could not be possible without the logistical and academic support of many people and organizations, so thanks to AUT for their support early on, and Massey University during the course of this PhD. Griffith University for the opportunity to undertake genetics lab work. To the International Sea Turtle Society (ISTS) and its members, as an independent and often isolated researcher, the opportunity to attend such great international symposiums to present my work and meet with inspiring and helpful scientists has been invaluable. It many ways it has certainly kept me going. To my fellow students and colleagues of the Coastal-Marine Research Group, thanks for the support and friendship, particularly to Krista Hupman (yes you beat me to it but I'm still the Don ☺), Sarah Dwyer, Sonja Clemens, Manue Martinez, Emma Betty, Cheryl Cross, Anna Meissner, Gabriel Machosvsky Capuska, Jochen Zaeschmar. To my collaborators, thanks for your contributions your help was invaluable and indeed beneficial to the quality and depth of the research. Adam and Mat, many thanks for the statistical analyses despite your own commitments. Ken Mackay, thanks for your interest in all things turtle down under. To Wendy Nelson (NIWA), Richard Willan Rick Webber (Te Papa Museum), Colin Mclay (Te Papa Museum), Clinton Duffy (DOC), Emma Betty (C-MRG) for identifying diet items and Brian Gill (ex-Auckland Museum) for sharing his historical data. Thanks to Michael Jensen (NOAA) for all the help with green turtle genetics and reference sequences. A very special thanks to my collaborators and mentors Dr. Colin Limpus and Dr. Nancy FitzSimmons, your continued support was invaluable. My deepest gratitude also goes to a friend and fellow scientist, Dr. David Feary who gave his time to review my drafts and provide great advice. Mate, you have made me a better scientist and pulled me out of so many writing holes! Those who volunteered endless hours with necropsies also deserve sincere thanks including Ale Bielli, Cat Lea, Blair Outhwaite, Zoe Lyle, and Joanna Crawford. I am also very grateful to the Institute of Natural and Marine Sciences for awarding me a Fee Scholarship, logistical support, and satellite tracking costs; AUT for initial research funding; Kelly Tarlton's for funding and supporting the satellite tracking programme; DOC for logistical support and permit waivers; for travel support to present at the PEW symposium; EMR and Dargaville Intermediate; travel grants from ISTS, the Zoological Society of Auckland, and Karen Baird (New Zealand Forest and Bird).

On a personal level, a huge amount of gratitude is owed to my friends far and wide. They have supported (and distracted me) over the years. I thank my immediate and extended family for all their support and endless belief in me despite my own uncertainties. I am forever grateful. In particular, to my dearest Cara, you are my rock, my safe anchorage, my endless source of inspiration and laughter, this work is just as much yours as it is mine. It quite literally is…who else would want to go on a date to necropsy smelly dead turtles and willingly read endless drafts!? ☺. Cara, I'm eternally grateful for your unwavering support.

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Chapter 1 **General introduction**

Successful conservation of threatened species requires an understanding of their biology, their relationship with the environment, as well as identifying and mitigating threats to their survival (Crowder and Norse 2005, Cooke 2008, Block et al. 2011). A fundamental aspect, therefore, is quantifying baseline information about the species' population structure, spatio-temporal distribution and habitat use. These key objectives toward ecological knowledge are critically important for long-lived and highly migratory marine species, and particularly for those species that are also under threat by anthropogenic impacts. In this context, also determining the genetic structure and bio-geographic connectivity of a population is an important facet of their conservation (Reece et al. 2005). Only once an adequate understanding of the fundamentals of a species' ecology is established, can conservation managers adequately investigate and identify human induced threats and develop appropriate mitigation policy. Therefore, without empirical baseline data, threatened species cannot be effectively managed to halt their decline.

The green turtle (*Chelonia mydas*) is a slow growing and long-lived marine reptile (Chaloupka and Limpus 1996, Limpus and Chaloupka 1997, Van Dam 1999). Its sexual maturity is delayed (Chaloupka and Musick 1997, Miller 1997); and breeding adults display strong natal homing behaviour and nest site fidelity (Meylan et al. 1990, Lohmann and Lohmann 1996, 1998). Green turtles exhibit complex life history patterns which are marked by distinct ontogenetic habitat shifts, coupled with long-distance breeding/foraging migrations (Bjorndal 1997, Bolten 2003). Consequently, a population's total range can be extensive – encompassing coastal nesting areas, epipelagic habitat, neritic foraging grounds, and migratory pathways (Balazs 1976, Bolten 2003, Luschi et al. 2003). These biological characteristics expose green turtles to a myriad of anthropogenic impacts, and as such, there has been a substantial decline at a global scale of this species (Wallace et al. 2011, Wallace et al. 2013, Lewison et al. 2014). Consequently, the green turtle is listed as globally 'Endangered' in the

International Union for the Conservation of Nature's Red List of Threatened Species (Seminoff 2004).

The green turtle has a circum-global distribution, ranging throughout tropical and subtropical seas between latitudes 40°N and 35°S (Màrquez 1990, Hirth 1997). Nesting occurs across a narrower range between 30°S and 30°N. Although considerable research has been undertaken on this species, most of our current knowledge relates to breeding adults at nesting grounds, and within its core preferred habitat of equatorial tropic and sub-tropical regions. Comparatively, very little is known of the distribution and migratory behaviour, foraging and habitat use of post-hatchlings through to sub-adult age classes. Indeed, empirical data are particularly limited for this species outside what is considered their preferred core habitat i.e. at latitudes 30°N and 25°S (Màrquez 1990).

Within New Zealand's Exclusive Economic Zone (EEZ), green turtles are infrequently bycaught in fisheries (Rowe 2009, CSP 2010, Ramm 2012a, 2012b, Clemens-seely et al. 2014), or occasionally found stranded along New Zealand's coastline (Gill 1997). Free-ranging green turtles are also reported intermittently within New Zealand's inshore waters, primarily off north-eastern New Zealand (Gill, 1997). Thus far, however, it is considered these individuals have arrived in New Zealand waters after becoming disoriented or blown off course by strong weather events during migration, arriving as stragglers from Australia or beyond after being dispersed outside their natural range (McCann 1966, Gill 1997). Furthermore, no study has examined the population structure (Godoy et al. 2016; Chapter 2), genetic origin and regional connectivity (Chapter 3), diet (Chapter 4), and anthropogenic impacts (Chapter 5), in New Zealand. Hence, this study aims to investigate these key biological aspects of the green turtle in New Zealand to provide a baseline understanding of this species at the edge of their range. This study will make an original contribution to the current body of knowledge, and ultimately, inform conservation management of this globally endangered species in New Zealand waters and the South Pacific region. This first chapter provides an overview of the green turtle relative to the objectives of this study.

1.1 Taxonomy

The green turtle was first described by Linnaeus (1758) as *Testudo mydas*. The generic name *Chelonia* was assigned later by Brongniart (1800). From a taxonomic-nomenclatural perspective, green turtles belong to the family Cheloniidae within the order Testudines. Of the seven extant species of marine turtle, all except one species (i.e. *Dermochelys coriacea*) fall into this clade. At a specific level, *C. mydas* has been the focus of considerable interest and research investigating the biogeographic, morphological, and biochemical divergence of this species (Carr 1975, Hirth 1997, Bowen and Karl 2007, Limpus 2008a). As such, several specific and subspecific names have been proposed. Within the Pacific and Indian Ocean basins, Carr (1975) proposed the separation of the green turtle into the eastern Pacific form *C. agassizi* (encompassing a region from Baja California to the Galapagos Islands and Peru, and westward to the Hawaiian Archipelago); and *C. japonica* (for the Indian Ocean and the western tropical Pacific). This bilinear designation followed on from Bocourt (1868), who first described *C. agassizii* as a distinct species. For the Atlantic Basin, Carr (1975) also proposed that *C. mydas mydas* be used for the Ascension Island population (south-equatorial mid-Atlantic); and that *C. mydas viridis* be used for the Tortuguero colony (Caribbean coast of Costa Rica).

More recently, the phylogeny of the green turtle has continued to be a matter of considerable interest and study. Proponents of the divergence of *Chelonia* into one or more distinct species or sub-species include Pritchard and Trebbau (1984) who recognised the East Pacific green turtle *C. agassizi* as a distinct species, yet acknowledged some degree of occasional sympatry with *C. mydas* in Pacific Mexico, the Galapagos and Papua New Guinea. Màrquez (1990) also recognised *C. agassizii* from the eastern Pacific, as well as suggesting *C. mydas* is comprised of two subspecies: *C. m. mydas* in the Atlantic Ocean and *C. m. japonica* in the Indian Ocean and in the western and central Pacific Ocean. It should be noted however, that these investigators based their findings on morphological diagnostic features, such as adult carapace vaulting and tapering, colouration and the geographic isolation of the studied populations (Carr 1975, Pritchard and Trebbau

1984, Màrquez 1990). The use of such characteristics alone has been shown to be problematic; particularly in regard to using colour as a single diagnostic tool. For example, Frazier (1971) found substantial colour variation in his study of a single population in Aldabra Atoll in the Indian Ocean. Karl and Bowen (1999) states “although the black turtle is morphologically identifiable at a low level, much of its distinction is based on size and color differences that are highly variable throughout the range of *C. mydas*”. Parham and Zug (1996) further recommended that the name *C. mydas* be used, with no formal subspecific recognition for green turtles throughout the world. Hirth (1997) supports this view, describing *C. mydas* as a “circumglobal morpho-species [...] made up of several distinct populations and metapopulations”. Bowen and Karl (2007) expand on this, postulating the divergence between the Atlantic and Indo-Pacific green turtle populations occurred several million years ago, hence warrant the recognition of “subspecific evolutionary entities”, yet these divergences “are not sufficient [enough] to generate reproductive isolation and speciation”. Today, *C. mydas* is formally recognised as a monotypic species with a worldwide distribution across tropical and sub-tropical seas (Hirth, 1997).

1.2 Morphology and identification

Green turtles are the largest of the cheloniids (hard-shelled turtles), with adults commonly attaining weights in excess of 150 kg and curved carapace lengths (CCL) >100 cm (Màrquez 1990, Hirth 1997, Pritchard and Mortimer 1999). In cheloniids, the main characteristics used for identification are the structure and arrangement of the scutes (scales) of the carapace (dorsal shell) and the presence/absence and number of prefrontal scales between the eyes (Figure 1b, 1c). The basic external structures of green turtles are outlined in Figure 1. Although size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics, all should be taken into consideration when distinguishing species and describing an individual. For reasons discussed earlier (e.g. Frazier, 1971), colour variation alone should not be considered diagnostic to distinguish between breeding populations.

The carapace of the green turtle is smooth, streamlined and high domed. The carapacial scutes are heavily keratinized, juxtaposed and do not overlap i.e. non-imbricate (Solomon et al. 1986, Hirth 1997, Limpus 2008a). Carapace scute arrangement consists of five vertebral scutes, four pairs of costal scutes, and twelve pairs of marginal scutes – including the supracaudals (Figure 1b). The scutes of the plastron are also heavily keratinized (Solomon et al. 1986) and comprise intergular, gulars, humerals, pectorals, abdominals, femorals, and anals (Hirth, 1997). The four enlarged inframarginal scutes lack pores. The skull has a single pair of prefrontal scales and usually four pairs of postocular scales (Figure 1c).

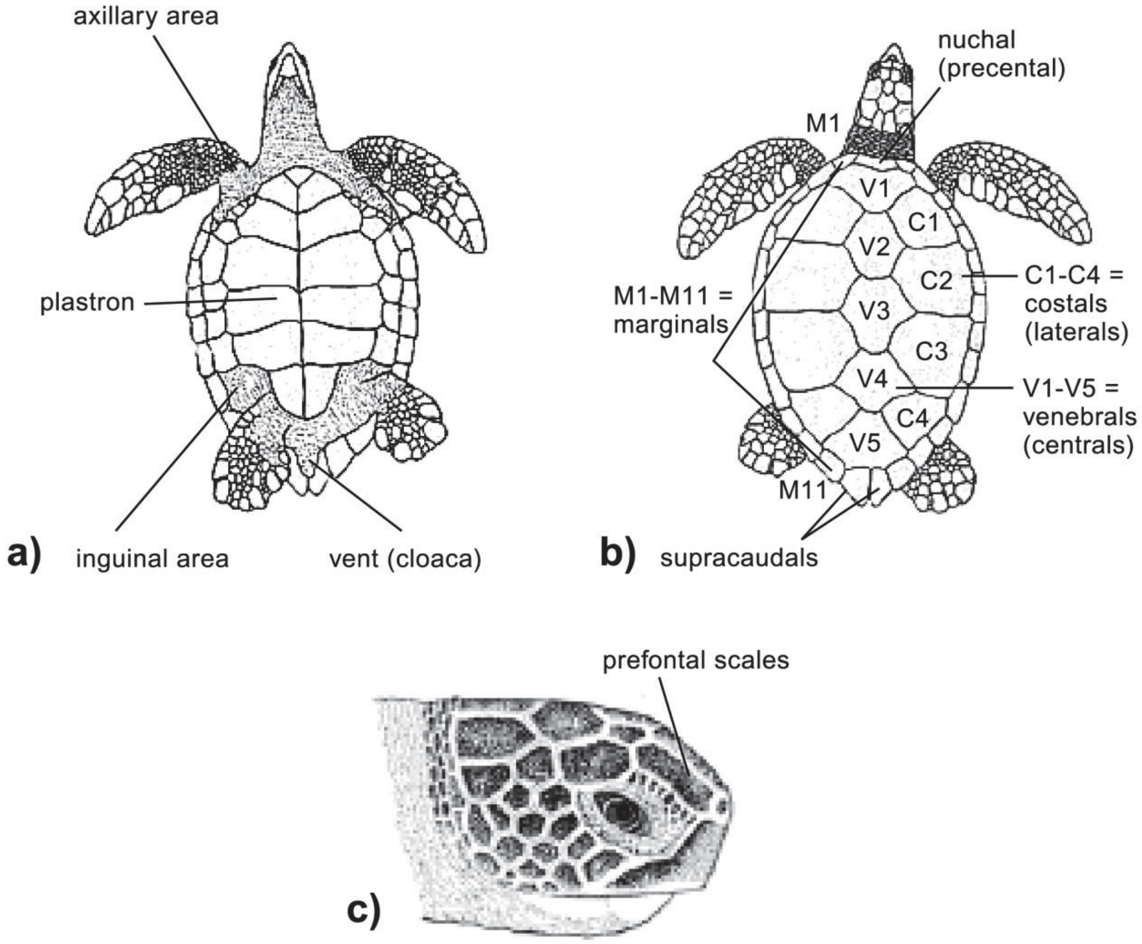

Figure 1. Ventral (a) and dorsal (b) view showing basic external structures of a Cheloniid marine turtle. Figure 1c shows magnified side-view of a green turtle head, showing prefrontal scales (source: Pritchard and Mortimer 1999).

The morphology of green turtles originating from east Pacific rookeries differs slightly from green turtles from other Pacific stocks. Overall, the east Pacific green turtle is smaller in size and darker in colour – hence sometimes referred to the black turtle. The posterior edge of the carapace (over the hind flippers) concaves inward, the carapace is higher domed anteriorly, and the plastron is blotched with dark spots (Màrquez 1990, Pritchard and Mortimer 1999).

Marine turtles do not exhibit external reproductive structures, and secondary sexual characteristics only develop in mature adults (Wyneken 2001, Phelan and Eckert 2006), thus, sexual dimorphism is only detectable in large sub-adults and adults. In a study of 12 populations, Godley et al. (2002) found a consistent pattern of sexual dimorphism where adult males were smaller than adult females. Smaller adult male size has been shown in other populations in the Pacific (e.g. Australia; Limpus and Chaloupka 1997) and the Atlantic basins (e.g. Brazil; Grossman et al. 2007). Other morphometric differences that can help determine sex in mature adults are carapace morphology, elongation and curvature of the claws on the fore-flippers of males, and the most conclusive characteristic being tail length in mature males (Pritchard and Mortimer 1999, Godley et al. 2002). To date, no investigation has been undertaken into the morphological characteristics of green turtles in New Zealand.

1.3 Growth and sexual maturity

Green turtles are slow-growing long-lived animals that undergo several distinct life stages as they grow and eventually reach sexual maturity (Chaloupka and Limpus 1996, Chaloupka and Musick 1997, Limpus and Chaloupka 1997, Van Dam 1999, Bolten 2003, Limpus et al. 2003). Many factors affect the growth rate of green turtles including an individual’s physiology, age, sex, diet and geographic location (Hirth, 1997). As such, growth rates and ultimately the age at which individuals reach sexual maturity can vary widely.

Growth rates for wild populations vary, yet are thought to be a function of size/maturity, the geographic location of the feeding ground, and the nutritional quality of the forage therein (Bjorndal et al. 2000, Limpus 2008a). Studies have shown that juveniles grow at higher rates than older age classes with growth in adults slowing when maturity is reached (Hirth, 1997). For example, immature turtles (small juveniles 20-30 cm straight carapace length; SCL) have exhibited growth rates of 3.6-9.0 cm/yr while in contrast, sub-adult turtles (80-90 cm SCL) have shown growth rates of 0.1-1.1 cm/yr. For the eastern Australian stocks, adult turtle (>95 cm CCL) growth is very slow and in some cases ceases all together (Limpus, 2008).

The age of sexual maturity varies considerably between populations, with estimates ranging between 26 and 40 years (Balazs et al. 1987, Limpus and Chaloupka 1997, Seminoff et al. 2002a, Koch et al. 2007, Senko et al. 2010). In Australia, sexual maturity has been estimated at between 30 to 40 years (Limpus and Walter 1980, Limpus 2008a). The mean reproductive age for nesting females for Heron Island, Australia may be about two decades (Limpus 2008a). Life expectancy for this species has been poorly quantified, however overall generation length has been estimated at ca. 35-50 years (Seminoff 2004).

In New Zealand, no study has investigated the age structure of the aggregation. However, Gill (1997) did determine the mean CCL of a small sample of stranded green turtles from New Zealand was 56.1 cm (*n* = 20), with a range of 41.5 cm – 90 cm.

1.4 Nesting

The reproductive behaviour of green turtles is complex; marked by long-distance breeding migrations and natal homing (Lohmann and Lohmann 2006). Green turtles have been recorded nesting in more than 136 countries worldwide, and with few exceptions, all nesting occurs within the latitudes 30° S and 30° N (Hirth, 1997). Predominantly, nesting occurs during the warmer summer months. In the southern hemisphere, this occurs primarily between October to March with a peak in nesting activity between December and January. Nesting at locations closer to the equator can occur all year round (Obermeier 2002, Yasuda et al. 2008). In the southwestern Pacific, large nesting aggregations occur on Australia's eastern seaboard, islands of the outer Great Barrier Reef, and islands further offshore in the Coral Sea (Limpus 2008a). Limpus (2008) suggests Raine Island in northeast Australia is the largest remaining green turtle nesting population in the world. Nesting also occurs in New Caledonia, Papua New Guinea, Solomon Islands and to lesser extent, eastward throughout Polynesia, where nesting is scattered across Tonga, Western and American Samoa, Tokelau and French Polynesia (Hirth 1971, 1997, Balazs 1983, Guinea 1993, Tuato'o-Bartley et al. 1993, Allen 2007). No nesting has been recorded in New Zealand. In the eastern Pacific, the main nesting rookeries are located on the Pacific coast of Mexico, Costa Rica and the Galapagos Islands (Ecuador) (Màrquez 1990).

The number of nesting females in any one population varies considerably between years (Limpus 2008a) and, in the case of eastern Australia, is influenced by the Southern Oscillation Index approximately two years before the breeding season (Limpus and Nicholls 1988, Limpus and Chaloupka 1997, Chaloupka et al. 2004), with males appearing to be regulated in a parallel manner (Limpus 2008a). The regulatory effect of this regional climatic event is most likely related to its effect on the quantity or nutritional quality of the diet (e.g. seagrass productivity). In turn, any nutritional deficiency experienced by mature adults will affect their overall body condition and reproductive fitness (Limpus 2008a). Due to the physiological demands of nesting on adults, both in regard to migration to and from nesting rookeries and production of clutches, adults do not breed in subsequent years. For the eastern Australian breeding stocks, nesting occurs usually every three years to six years (Limpus 2008a).

1.5 Range, dispersal and distribution

The green turtle has a circum-global distribution, ranging throughout tropical and subtropical seas between the latitudes 40° N and 35° S (Màrquez 1990, Hirth 1997). Given the complex life history of this species, exemplified by distinct ontogenetic habitat shifts throughout their life time, green turtles occupy a range of geographical and temporal habitats. In addition, ocean currents play an important role in the dispersal and aggregation of marine turtles, thus introducing a degree of stochasticity to their distribution (Carr 1980, Bolten 2003, Luschi et al. 2003, Bass et al. 2006, Dethmers et al. 2010). The differential distribution of green turtles begins in neonate hatchlings; the life stage encapsulated from when neonate green turtles emerge from the nest until the turtle begins to actively forage for food (Musick and Limpus 1997, Boyle and Limpus 2008). This is usually only a period of days while the hatchling does not feed but survives on the yolk sac which is retained within the body cavity (Limpus 2008a). Once hatchlings emerge from the nest, they orient themselves toward the low level light horizon of the sea (Limpus 1971, Lohmann and Lohmann 1996). When they enter the water they begin what is commonly referred to as a ‘frenzied swim’ phase, keeping a constant compass course that takes them away from the coast by swimming perpendicular to wave fronts (Lohmann and Lohmann 1996, 2010). This frenzied swim episode continues for more than 24 hours and has been inferred as a strategy to quickly leave predator-abundant shallow areas near the coast to deeper waters (Gyuris 1994, Salmon et al. 2009). Following this period of frenzied swimming, the degree to which post-hatchlings undertake self-directed migrations is poorly understood. The likelihood is that this age class disperses passively (rather than actively migrates) into offshore oceanic gyre systems completing a period of development in epi-pelagic habitats as surface dwelling planktivores (Lohmann 1992, Salmon and Wyneken 1994, Boyle 2006, Limpus 2008a). Originally known as the ‘lost year’, this period is more appropriately termed ‘lost years’.

In the South Pacific Ocean hatchling dispersal from nesting rookeries, as well as the dispersal of juveniles and adults, is influenced by oceanic factors associated with the anti-cyclonic South Pacific Subtropical Gyre (SPSG) (Limpus et al. 1994, Boyle and Limpus 2008, Boyle et al. 2009, Pendoley and Christian 2012). For example, tag and data logger recoveries (e.g. passive integrated transponder (PIT) tags, flipper tags, and satellite tags) from juvenile and breeding adults reveal a generalised westward migration/dispersal pattern which is consistent with the direction of the South Equatorial Current (SEC) (Trevor 2010, Pendoley and Christian 2012). Marine turtles dispersing or migrating via the SEC may eventually enter the Coral Sea by zonal surface jets, then be carried southward down the East Australian Current (EAC), and potentially eastward into the Tasman Front (TF) (Figure 2). Pendoley & Christian (2012) for example, found a juvenile green turtle that had been tagged and released in Tahiti, on 1 October 1993, was eventually found at Norfolk Island, 17 months later on 4 March 1995.

Furthermore, hatchlings emerging from eastern Australian nesting rookeries are also considered to passively disperse via these same sea surface currents by way of the EAC into the western boundary of the South Pacific Subtropical Gyre (Boyle and Limpus 2008). It is in this anti-cyclonic gyre system and associated eddies, where it is believed they may reside for several years in the epi-pelagic zone. Stranding records for eastern Australia indicate post-hatchling green turtles are swept southward from southern Great Barrier Reef (sGBR) waters by the EAC, marginally past New South Wales and then out into the Pacific Ocean via the Tasman Front (Limpus et al. 1994, Walker 1994).

After a period of 5 to 10 years (at approximately 40 cm CCL), juveniles leave the oceanic habit and recruit into neritic developmental habitats until reaching maturity some decades later (Zug and Glor 1998, Zug et al. 2002, Limpus 2008a). For instance, for the Heron Island breeding stock, age at recruitment is approximately 10 years (Limpus et al. 2005). The primary mechanisms that influence the timing of recruitment and habitat selection is largely unknown, yet is most likely a response to changes in their nutritional and habitat requirements (Bjorndal 1997, Musick and Limpus 1997).

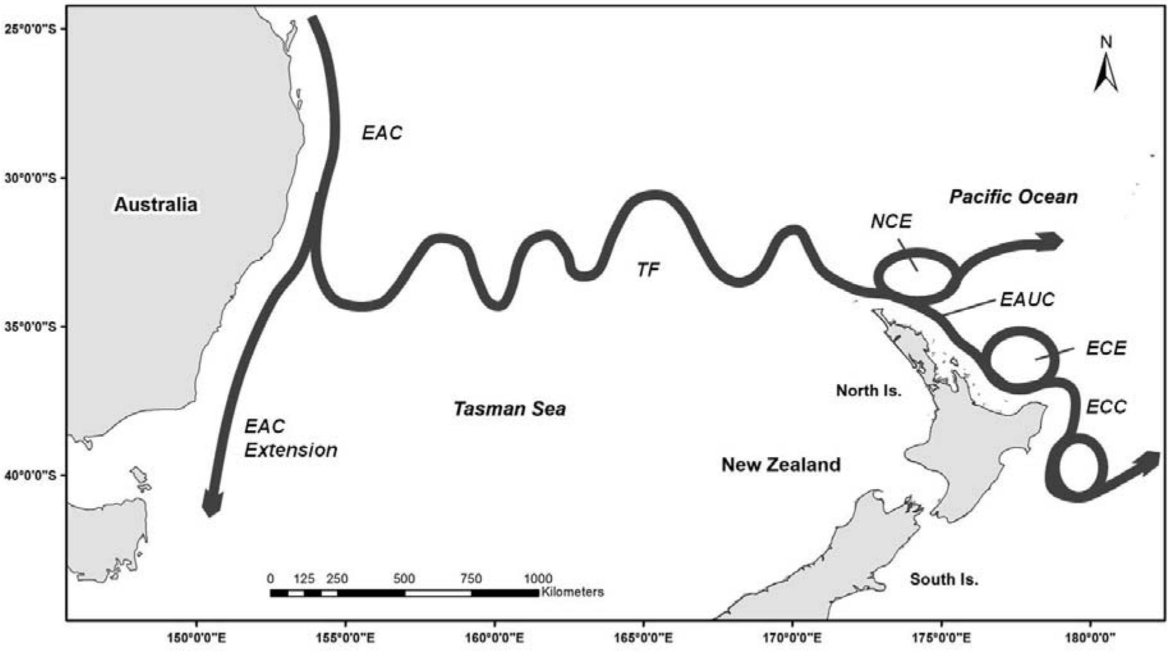

Figure 2. Generalised diagram of the main sea surface currents and associated eddies of the southwestern boundary of the South Pacific subtropical gyre system on the east coast of Australia and northeastern New Zealand: East Australian Current (EAC), Tasman Front (TF), North Cape Eddy (NCE), East Auckland Current (EAUC), East Cape Eddy (ECE), and East Cape Current (ECC) (modified from Tilburg et al. 2001).

In mature adults, it has been demonstrated that both sexes exhibit strong natal homing behaviour, where they repeatedly return (following periodic foraging migrations) throughout their reproductive lives to the same area from where they originally hatched, to mate and nest. Tag returns from females tagged at northern Great Barrier Reef (nGBR) rookeries have been recovered as far away as the Gulf of Carpentaria, eastern Indonesia (Aru, Ambon, Kei, Irian Jaya), southern and eastern Papua New Guinea, Vanuatu, New Caledonia, and southern Queensland (Limpus 2008a), indicating the extent of their foraging migrations and preferred foraging grounds. Internesting migrations have been extensively studied, which show adults will often migrate up to thousands of kilometres between nesting sites and preferred foraging grounds. These migrations often occur across vast expanses of open ocean and frequently in a linear manner (Lohmann et al. 1999). This navigational ability in marine turtles is poorly understood, however, evidence suggests it is during hatching, that neonate turtles are imprinted to the inclination of the earth's magnetic field at the nesting beach (Lohmann 1991, Light et al.

1993, Lohmann et al. 2008), hence are able to return to the same geographical area to breed many decades later.

Given that New Zealand lies at the southwestern extremity of the Polynesian triangle (Figure 3), its landmass dissects the same area of the western boundary of the South Pacific subtropical gyre described previously. Its territorial waters span a considerable latitudinal range, encompassing the sub-tropical regions of the Kermadec Islands to the north (~30°S) down to the sub-Antarctic Islands at ~51°S (Figure 3). Therefore, it is plausible to infer that green turtles dispersing either initially westward (via the SEC) and then down into the EAC, or, eastward from Australian rookeries, could become entrained within the Tasman Front, eventually getting swept toward New Zealand. If so, it would be expected to occasionally encounter oceanic phase green turtles in New Zealand waters. In contrast, Gill (1997) did not report small juvenile green turtles (i.e. <40 cm CCL), yet did find green turtles in the range of 41.5 cm – 90 cm CCL (μ = 56.1 cm; n = 20). Although a very small sample size, this implies that green turtles encountered in New Zealand comprise post-oceanic juveniles through to sub-adult/adult age classes. However, given the small sample reported by Gill (1997), further investigation would provide a better understanding of the dispersal, migratory pathways and post-oceanic settlement of green turtles in this region of the southwestern Pacific.

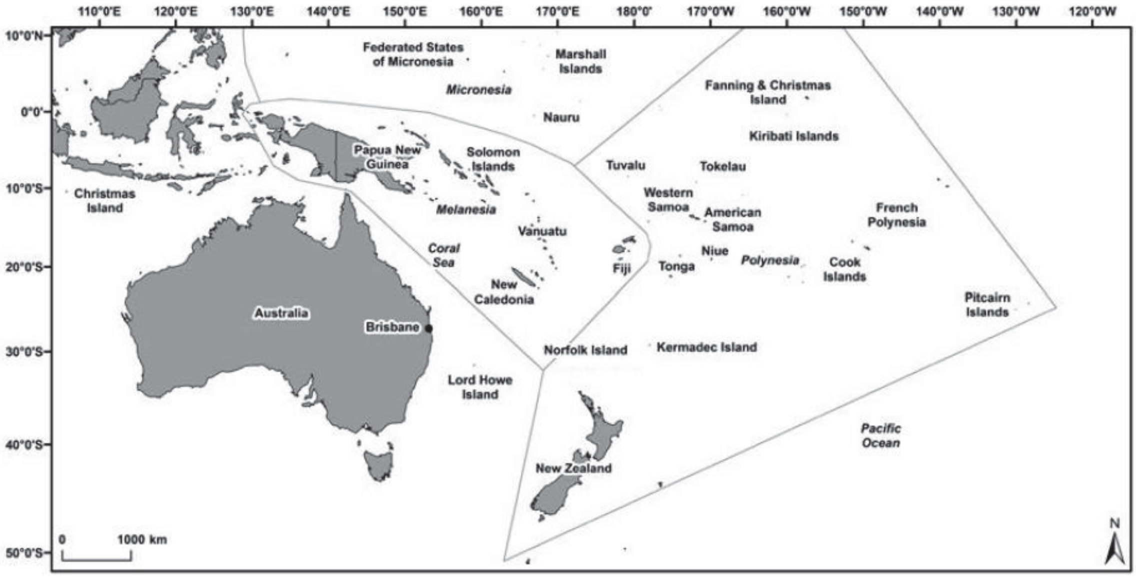

Figure 3. New Zealand lies at the southwestern extremity of the Polynesian triangle, and encompasses the sub-tropical regions of the Kermadec Islands to the north down to the sub-Antarctic Islands in the south (from Pendoley and Christian 2012).

1.6 Phylogeography

Understanding population genetics of threatened species has become a critical facet of conservation in recent decades (Avise 1998). The utility of molecular tools have proven particularly beneficial for the study and management of marine species because, in comparison to their terrestrial counterparts, delineating population boundaries of often highly migratory and widely dispersed species poses significant challenges (Witteveen et al. 2004, Jorgensen et al. 2010, Carvalho et al. 2014). With regard to marine turtles, the development of molecular analysis techniques has enabled more explicit investigations of genetic divergence between nesting colonies. Since mitochondrial DNA (mtDNA) is maternally inherited, molecular evolution and matriarchal lineages could be investigated. Early studies analysed mtDNA restriction sites to identify differences in genotype frequency between geographically separate rookeries (Meylan et al. 1990, Bowen et al. 1992).

In a study of four Atlantic Ocean and Caribbean Sea nesting colonies, Meylan et al (1990) reported sufficient genetic distinctiveness between rookeries to suggest low levels of female-mediated gene flow. In turn, their study supported Carr's (1975) natal homing hypothesis where mature female green turtles return to their natal nesting area to breed (Meylan et al. 1990). Expanding this, Bowen et al (1992) also used mtDNA restriction sites to investigate the global population structure of the green turtle in terms of matriarchal phylogeny. They found a clear phylogenetic split between the Atlantic and Pacific Ocean Basins, low mtDNA variability overall and low mtDNA evolutionary rate compared to other vertebrates. The global phylogeographic divergence observed was posited as a result of geography (land-mass barriers of southern Africa and South America) and natal homing behaviour.

As molecular techniques progressed researchers were able to identify and isolate population specific genomic sequences at the control region of mtDNA using species targeted primers (Avise and Bowen 1994, Norman et al. 1994). Initially, these primers isolated a ~380-480 bp sequence amplified from the 5' end of the control region. Because the control region evolves faster and contains significantly more characteristic variation than other regions of the genome (Moritz et al. 1987), analyses of these segments yield higher genotypic resolution than can be obtained from analyzing nucleic restriction fragment length polymorphisms (RFLP). Norman et al (1994) demonstrated this, whereby they were able to discover a higher degree of genotypic structuring of Indo-Pacific green turtle stocks than previously identified using RFLP analysis. The discovery of genetically discrete breeding stocks occupying the same regions revealed the demographically independent nature of marine turtle populations despite their proximity; hence, illustrating the need to manage them as discrete biogeographical management units (Bowen et al. 1992, Norman et al. 1994).

Although a ~380-480 bp sequence analysis can adequately differentiate between proximate rookeries (thereby identifying discrete breeding stocks), they are limited in their resolution of stocks at foraging grounds where multiple stocks converge (Abreu-Grobois et al. 2006). Addressing this, novel primers have allowed the amplification of longer sequences (~800 bp) which provide suitable resolution between stocks at foraging and developmental grounds (Abreu-Grobois et al. 2006). Understanding this aspect of their biogeographical distribution is important for wildlife managers in order to delimit specific management units that encompass the entire range and all life stages of the threatened population so that targeted conservation measures can be employed (Encalada et al. 1996, Wallace et al. 2010). Identifying and monitoring populations at foraging grounds of mixed origin stock has become an prominently critical facet of conservation management of these species at a regional level (Encalada et al. 1996, Abreu-Grobois et al. 2006).

In New Zealand, no study has investigated the genetic origin of the green turtle population and therefore the biogeographic connectivity to regional breeding stocks. For the purpose of this study, I follow Bowen and Karl (2007) and Moritz (1994), in recognizing *C. mydas* as a monotypic species found worldwide.

1.7 Diet and foraging behaviour

Diet and foraging behaviour are inextricably linked to an organism's life history strategy, distribution and movement, and ultimately reproduction and survival (Monteith et al. 2014, Resano-Mayor et al. 2016). Growth and reproductive output in marine turtles is directly influenced by nutritional resource acquisition (Balazs and Chaloupka 2004, Arthur and Balazs 2008), therefore understanding their diet and foraging ecology is an important component of conservation management (Bjorndal 1997). For green turtles, their diet is closely associated with the spatial and temporal habitat use and ontogenetic habitat shifts experienced during their life cycle (Bjorndal 1997). As post-hatchlings occupying open ocean habitats, green turtles most likely forage as opportunistic omnivores on macro-plankton in the epi-pelagic zone (Bjorndal 1997, Boyle and Limpus 2008). Once juveniles recruit and take up residence in neritic habitats, they transition to a benthic foraging strategy feeding year round in nearshore tidal and subtidal areas, sea grass meadows, algal turfs, and coral and rocky reefs (Limpus 2008a).

It is considered that at this post-settlement stage they become primarily herbivorous for the remainder of their lives (Limpus 2008a), feeding primarily on seagrass, and a wide range of algae and mangrove fruits (Lanyon et al. 1989, Read 1991, Forbes 1996, Read et al. 1996, Limpus 2008a). To facilitate this dietary transition from omnivory to herbivory, a physiological shift in digestive function of the gut occurs (Bjorndal 1980). This process leads to a change in the composition and specificity of gut microflora which allows for hindgut fermentation of seagrass and macroalgae to produce volatile fatty acids (VFA) for energy (Bjorndal 1979, 1980). In order to maximise digestive efficiency, this ontogenetic shift is posited to occur abruptly and irreversibly (e.g. Bjorndal 1979, 1980, Reich et al. 2007).

More recently, studies across its range challenge the paradigm that green turtles transition to a strictly herbivorous diet once they recruit into nearshore developmental grounds. There are a number of studies demonstrating that green turtles continue to supplement their diet with animal prey during this phase and that the transition is less definitive than previously posited (Hatase et al. 2006, Cardona et al. 2009, González et al. 2012, e.g. Morais et al. 2014). The variability of the diet is most likely influenced by a number of factors such as the spatio-temporal dynamics of food availability (Bjorndal 1980, Garnett et al. 1985, Forbes 1996, Brand-Gardner et al. 1999, González et al. 2012), composition of gut microflora (Bjorndal 1979, Bjorndal et al. 1991), an individual's nutritional needs and foraging strategy (Forbes 1996, Amorocho and Reina 2007, 2008, Vander Zanden et al. 2013), environmental perturbation (Gama et al. 2016) and predation risk (Burkholder et al. 2011, Meylan et al. 2011). In particular, because green turtles are poikilothermic, ambient environmental temperature affects digestion rates and thus digestive efficiency (Bjorndal 1980, Mrosovsky 1980, Spotila and Standora 1985, Spotila et al. 1997). As a consequence, lower ambient temperatures are likely to influence diet selection and ingestion rates (Bjorndal 1980, Mendonca 1983, Balazs et al. 1987, Amorocho and Reina 2008, Reisser et al. 2013, Morais et al. 2014). This aspect has considerable implications for green turtles that are found at the edge of their range in temperate habitats such as New

Zealand. As yet, no investigation has been undertaken to determine if green turtles found in New Zealand are foraging and if so, what the components of the diet are.

1.8 Anthropogenic Impacts

Human activities have had a dramatic effect on taxa worldwide which has led to a significant decline in biodiversity globally (Pereira et al. 2012). In the marine environment, key threats to species include overexploitation and harvesting, bycatch, habitat loss and degradation, and pollution (Lewison et al. 2004, Shillinger et al. 2008, Block et al. 2011, Gilman 2011). In recent decades, climate change and plastic pollution have emerged as significant drivers of environmental change and biodiversity loss at a global scale (Derraik 2002, Burrows et al. 2011). In relation to marine taxa, long-lived migratory species such as sharks, marine mammals, seabirds and marine turtles are particularly affected by human effects because they are often concurrently exposed to many cumulative impacts throughout their lives (Block et al. 2011). In addition, their life history traits lead to slow population recovery following decline (Lewison et al. 2004). In order to understand and mitigate for anthropogenic impacts, conservation managers need to initially identify species and population levels threats before appropriate strategies can be developed (Block et al. 2011).

The green turtle, like all marine turtle species, is vulnerable to anthropogenic related decline, and although historically abundant, human over-exploitation, habitat loss and pollution have caused significant declines in abundance of this species worldwide (Seminoff 2004). Human induced impacts affect every stage of a green turtle’s life cycle – from eggs through to mature adults. These impacts can be broadly categorised into direct (e.g. exploitation and incidental catch) or indirect effects (e.g. habitat loss, climate change and pollution). The cumulative effects of human activities on green turtle populations have shown significant negative impacts across all ocean basins (Bugoni et al. 2001, Labrada-Martagón et al. 2011, Clarke et al. 2014, González, Acha, et al. 2014, Pilcher et al. 2015, Laffoley and Baxter 2016, Schuyler et al. 2016). Despite the breadth of research internationally, in New Zealand, no study has investigated the potential effects of human activities upon green turtles in the region.

1.8.1 Fisheries interactions

In recent decades, the industrialisation of the commercial fishing sector has led to significant declines in non-target and protected taxa globally (Lewison et al. 2004, Block et al. 2011). Correspondingly, mortalities at sea through incidental catch from commercial fishing practices have been demonstrated to have a significant impact on marine turtle populations worldwide (Wallace et al. 2013, Lewison et al. 2014). Commercial fishery methods implicated in green turtle bycatch include trawl, seine, set net, and demersal and surface longline activities (Màrquez 1990, Hillestad et al. 1982). Although marine turtle bycatch is often underreported and inherently difficult to estimate accurately, in the southwest Pacific for example, trawl fisheries have been identified as a significant cause of decline in the eastern Australian marine turtle stocks (Limpus 2008a). In response, Turtle Excluder Devices (TED) have been used with a degree of success in Australian fisheries. Boat strikes (although not isolated to the fishing industry) have also been shown to pose to a risk to marine turtles (Gardner et al. 2003, 2006, Orós et al. 2005, Work et al. 2010, Labrada-Martagón et al. 2011, Parra et al. 2011, Denkinger et al. 2013).

In New Zealand, information on fisheries interactions with marine turtles is limited, therefore, the impact of fisheries activities on green turtles in New Zealand is poorly understood (Godoy 2016). In his 1997 review, Gill found 41% (n = 28) of marine turtles (four species combined) had been entangled by fishing gear (nets and lines). The study did not state whether these were incidental captures from commercial or recreational fishing activities. Between 2001 and 2010, the New Zealand government's Conservation Services Programme (CSP) commercial fisheries observer programme has reported very low level of green turtle interactions across the entire commercial fleet (Harley and Kendrik 2006, Brouwer and Griggs 2009, Godoy 2016). All incidental captures were from north-eastern North Island. Two green turtles were caught in surface longline fishery (Harley and Kendrik 2006), while another green turtle was incidentally captured in the inshore demersal/bottom longline snapper (*Pagrus auratus*) fishery (CSP 2010). A third record was reported from the inshore trawl fishery targeting John Dory; *Zeus faber* (Abraham and Thompson 2011). All four turtles were captured and reported to have been released alive. Given the low level of reported captures, it has not been possible to estimate statistically robust capture rates of green turtles in New Zealand (Harley and Kendrik 2006).

1.8.2 Pollution

Marine pollution has been defined as any substance or energy that has been directly or indirectly introduced into the marine environment by humans that causes deleterious effect to marine life (Islam and Tanaka 2004). Accordingly, it is a hazard or hindrance to human health or activities, and causes a reduction in the quality of the sea as an amenity. Therefore, there are a myriad of potentially harmful sources of marine pollution which pose a significant threat to the marine ecosystem and species worldwide (Engler 2012, Pereira et al. 2012). Marine debris, organic and inorganic pollutants, anthropogenic noise and light pollution are some of the key factors that have negatively affected species such as seabirds (Nevins et al. 2005, Finger et al. 2015), cetaceans (Boren et al. 2006, Unger et al. 2016), fishes (Carson 2013) and marine turtles (Nelms et al. 2016).

Marine debris poses a significant threat to green turtles, and is well documented in the literature (Carr 1987, Bjorndal et al. 1994, Bugoni et al. 2001). Green turtles are highly susceptible to ingesting marine debris mistaken as forage or prey items. As such, marine debris is directly ingested by hatchlings, juveniles and adults across the entire distributional range of this species. More recently, considerable interest has been focused on the impact of chemical pollution such as heavy metals (Gardner et al. 2003, 2006, Labrada-Martagón et al. 2011). Compounding this is the entanglement of green turtles in lost or discarded nets and other associated material, as well as non non-fishing related items (Oravetz 1999).

Light pollution at nesting sites has also been shown to significantly deter adult females from nesting and disorientate hatchlings as they emerge from the nest (Witherington and Martin 2000). Being positively phototactic, hatchlings during this critical emergence period can become disoriented, moving toward brighter landward light sources, rather than the desired low level sea-sky light horizon. Consequently, hatchlings moving inland and away from the sea quickly become exhausted, prone to predation, and usually die in the process (Witherington and Martin 2000).

1.9 Conservation status

Globally, the green turtle is listed as ‘Endangered’ in the International Union for the Conservation of Nature’s Red List of Threatened Species (Seminoff 2004). This assessment is based on a global review at the species level, stating: “extensive subpopulation declines in all major ocean basins over the last three generations...at 32 Index Sites distributed globally” (Seminoff 2004). As such, this species has been subject of various international legislative agreements and conventions to promote and facilitate conservation measures to halt the observed decline and restore populations to sustainable levels. These include the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Bonn Convention of Migratory Species (CMS), both of which New Zealand is signatory to.

Regionally, the Secretariat of the Pacific Regional Environment Programme (SPREP) administers the Regional Marine Turtle Conservation Programme (RMTCP) (SPREP 2012). New Zealand is a signatory nation to this programme. The RMTCP sets out objectives and activities for stakeholder nations to accomplish so that suitable conservation and management of green turtles is achieved in the Pacific. The RMTCP follows the recognition that green turtle populations should be managed as genetically discrete units, as recommended by Bowen et al. (1992), Bowen and Karl (2007), Moritz (1994), Norman (1994), and others. Currently, no study to date has investigated the genetic origin of green turtles in New Zealand or the regional connectivity of this wide ranging species.

Under the New Zealand Threat Classification System, the green turtle is listed as a *Migrant* (non-breeding) species (Hitchmough et al. 2010). In New Zealand, marine turtles are protected Under the New Zealand Wildlife Management Act (1953), and as such DOC is responsible for the conservation management of these species. However, without biological information underpinned by empirical data, developing informed and appropriate management plans or policies cannot be achieved. Nationally, current standard operating procedures for the management of marine turtles are limited and fragmented. In 2007, in recognition of the paucity of appropriate stranding response procedures for marine turtles in New Zealand, Northland's DOC conservancy commissioned an advice report to provide field staff with suitable handling and recovery procedures for stranded marine turtles (Godoy 2007: Appendix 1). Although implemented across Northland with some success, these procedures were not adopted nationally, thus, recovery efforts remain fragmented and ad hoc in other parts of the country.

1.10 Thesis outline

The objectives of this proposed research programme are to investigate the spatio-temporal distribution, population structure, genetic origin, regional connectivity, and anthropogenic impacts of green turtles in New Zealand. As yet, no research of this type has been undertaken in New Zealand; therefore, the data collected will be considered original work and will contribute to the international body of knowledge of this species. There are both practical and theoretical reasons for conducting this research. From a practical perspective, understanding the true extent and habitat use of this species across its geographical range will enable conservation managers to develop informed conservation strategies. This is particularly relevant for this species given its broad geographical distribution, where any one breeding stock may occupy a range of different habitats across international boundaries. Hence, this species often requires collaborative management policies between affected nations.

From a theoretical perspective, understanding the spatio-temporal distribution and habitat use of this species in habitats that are generally considered outside their natural range, would extend our knowledge of this species within the context of climate change and behavioural evolution. Ultimately, this research will advance our knowledge of the biology of green turtles in New Zealand waters, providing important biological data necessary for management of these IUCN Red Listed species and their habitat, eventually leading towards development of more appropriate international conservation strategies for this globally endangered marine reptile.

1.11 Thesis structure

The thesis comprises four research chapters (Chapters 2 to 5), with an introductory and concluding discussion chapter (Chapters 1 and 6). Each research chapter has been written in a publication format and represents a manuscript that is published (Chapter 2), submitted (Chapter 5) or in preparation for publication (Chapters 3 and 4). The publication status of each chapter is included herein. The aims of each chapter are outlined as follows:

Chapter One: Introduces the species with relevance to New Zealand, providing an overview of current knowledge and gaps in our understanding, as it relates to the distribution and habitat use of green turtles in the southwestern Pacific.

Chapter Two: Re-examines historical data in conjunction with recent stranding and sighting data of green turtles in New Zealand. Occurrence and necropsy data were analysed to determine the spatio-temporal distribution and population structure of the New Zealand aggregation. Historical sighting and stranding data (1895-2002) was provided by the Department of Conservation (Herpetological database) and B. Gill (Auckland Museum); while additional data (2002-2013) was collected by D. A. Godoy as part of this study. Spatio-temporal occurrence data was analysed by D. Godoy with Generalised Linear Model analyses (GLM) conducted in association with co-author A. N. H. Smith. Population structure data (biometrics, sex and maturity status) were collected during necropsy of stranded turtles and analysed by D. A. Godoy. The manuscript for this chapter was written by D. Godoy and improved by edits and suggestions by A. N. H. Smith, K. A. Stockin, C. J. Limpus, and D. A. Feary. This chapter is a reformatted version of a paper published as:

Godoy, D., Smith, A., Limpus, C. J., & Stockin, K.A (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), 549-565.

Chapter Three: Investigates the genetic origin and regional connectivity of the green turtle aggregation in New Zealand waters. Skin samples from live and dead green turtles in New Zealand were collected and catalogued under permit by D. Godoy between 2006 and 2014. Genetic samples were processed and mitochondrial DNA (~800 base pair fragment) was extracted by D. A. Godoy and co-author N. N. FitzSimmons (Griffith University, Australia). Nucleotide sequence analysis was conducted by D. A. Godoy and co-author N. N. FitzSimmons. Reference haplotype sequences were compared to published sequences and unpublished sequences provided by M. P. Jensen (US National Marine Fisheries Service - Southwest Fisheries Science Centre). The manuscript was jointly written by D. Godoy and N. N. FitzSimmons, and improved by edits and suggestions by K. A. Stockin. This chapter is currently in preparation for submission:

Godoy, D., and FitzSimmons, N. N. Connectivity across the Pacific: Origins of green turtles (*Chelonia mydas*) foraging in New Zealand waters.

Chapter Four: Examines the gut contents of stranded and incidentally caught green turtles to describe the diet of this species in New Zealand. Gut content samples were collected by D. Godoy during necropsy. Diet components were identified by D. Godoy with assistance of W. Nelson, R. Willan, B. Marshall, and R. Taylor. Analysis was conducted primarily by D. Godoy with assistance from M. D. M. Pawley. The manuscript for this chapter was written by D. Godoy and improved by edits and suggestions by K. A. Stockin and M. D. M. Pawley. This chapter is currently in preparation for submission:

Godoy, D., Pawley, M. D. M., & Stockin, K.A. The diet of green turtles (*Chelonia mydas*) at a temperate foraging ground of the southwestern Pacific.

Chapter Five: Investigates the anthropogenic impacts green turtles are exposed to in New Zealand waters. Data was collected by D. Godoy during necropsy of green turtles. Analyses were conducted by D. Godoy. The manuscript for this chapter was written by D. Godoy and improved by edits and suggestions by K. A. Stockin and D. A. Feary. This chapter is currently in preparation for submission:

Godoy, D. Anthropogenic impacts on green turtles (*Chelonia mydas*) in New Zealand.

Chapter Six: Summarises the key findings of this study with respect to the ecology of green turtles in New Zealand waters. The significance and contribution of the research to our understanding of green turtle ecology in the region is discussed, with limitations and future research outlined. Lastly, management considerations are proposed. This chapter was written by D. Godoy and improved by edits and suggestions by K. A. Stockin.

Chapter 2 **The spatio-temporal distribution and population structure of green turtles (*Chelonia mydas*) in New Zealand**

2.1 Abstract

Despite being endangered internationally and protected nationally, little consideration has been given to the occurrence of green turtles (*Chelonia mydas*) in New Zealand. New Zealand lies on the southern boundary of the distributional range of green turtles in the southwestern Pacific, with individuals found within these waters historically considered to be occasional visitors or stragglers incidentally carried by ocean currents. However, the present work shows that green turtles are present year round in New Zealand's northern waters (ca. 34°-38° S). A review of sighting, stranding, and incidental capture data collected between 1895 and 2013 illustrate New Zealand's green turtle population comprises post-pelagic immature juveniles to large sub-adults. The female:male sex ratio of 1.7:1 is similar to those reported from warm temperate foraging grounds in eastern Australia. A sub-sample of new recruits indicate green turtles recruit to neritic habitats at approximately 40.8 cm Curved Carapace Length. This study suggests that New Zealand's neritic habitats constitute a transitional developmental ground for post-pelagic immature green turtles. An exponential increase in the number of documented records over time was observed, though whether this is due to increased numbers of turtles or increased reporting rates, or both, is unclear and warrants further investigation. More broadly, this work provides a baseline understanding of the ecology of green turtles at the edge of their range, providing opportunities to investigate regional niche modelling and connectivity of this highly mobile species, while also monitoring broad-scale effects of climate-induced environmental change.

2.2 Introduction

The green turtle (*Chelonia mydas*) has a circum-global distribution, ranging throughout tropical and subtropical seas (Hirth 1997). They exhibit complex life history patterns which are marked by distinct ontogenetic habitat shifts, coupled with long-distance breeding and foraging migrations (Bjorndal 1997; Bolten 2003). Accordingly, a population's total range can be extensive – dispersing widely to encompass coastal nesting areas, neritic foraging grounds, oceanic habitats, and migratory pathways (Balazs 1976, Hirth 1997, Lohmann and Lohmann 1998, Bolten 2003, Luschi et al. 2003, Boyle and Limpus 2008). Upon entering the ocean for the first time, neonate hatchlings immediately disperse away from shallow predator-rich coastal waters into oceanic habitats where they forage as opportunistic omnivorous macro-planktivores in the epipelagic zone (Bjorndal 1997; Boyle and Limpus 2008). During this life stage, they will remain in the oceanic habitat for a period of 3 to 10 years, before settling into neritic foraging and developmental habitats at approximately 30–45 cm Curved Carapace Length (CCL) (Balazs 1985, Musick and Limpus 1997, Zug et al. 2002, Bolten 2003, Limpus et al. 2005, Arthur et al. 2008). Once juveniles recruit and take up residence in neritic habitats, they may remain geographically localised, possibly for decades until they reach maturity (Balazs et al. 1987, Limpus and Chaloupka 1997, Seminoff et al. 2002a, Koch et al. 2007, Senko et al. 2010). From the onset of maturity, adult green turtles undertake repeated breeding migrations to nesting rookeries, eventually returning to preferred foraging grounds in between breeding episodes (Hirth 1997).

Despite a wide global distribution, the geographic range of green turtles is generally constrained by the sub-tropical 20°C surface isotherm (Màrquez 1990, Davenport 1997, Hirth 1997). This is because, as with many marine ectotherms, marine turtles rely on the ambient environmental temperature to support digestion rates and, ultimately growth and survival (Avery et al. 1993). Therefore, temperature is one of the key environmental factors that regulate the physiological performance and ultimately the spatio-temporal distribution of green turtles (Spotila et al. 1997). If exposed to temperatures below their thermal tolerance for extended periods, adverse effects can lead to pathological and potentially fatal conditions (Schwartz 1978, Witherington and Ehrhart 1989, Spotila et al. 1997, Sadove and Pisciotta 1998, Southwood et al. 2003). Rapid decreases in temperature to below 10°C can result in 'cold stunning' and potentially stranding, while the lethal temperature for chelonid marine turtles in experimental conditions is ca. 8 °C (Schwartz 1978, Ogren and McVea 1995).

Temperate New Zealand is one habitat that falls outside the preferred thermal envelope (at least seasonally) for this species; where the average sea surface temperature (SST) in northern New Zealand ranges between 20-23 °C to 14 °C austral summer and winter, respectively (Chiswell 1994, Duffy 2002). Early published reports considered green turtles found in New Zealand waters as 'waifs or strays' or occasional visitors (McCann 1966). The most recent review of green turtle occurrence in New Zealand waters (Gill 1997), reviewed sparse sighting and stranding data spanning 125 years and reported an absence of green turtles in austral winter (July) and a significant bi-modal seasonal peak in abundance (austral summer and spring). Although this review did acknowledge that a small sample size and seasonal observer bias limited the strength of the results, this work concluded that green turtles were 'stragglers' from Australia or further north (Gill 1997). Green turtles were mostly observed during warmer seasonal months within northern New Zealand waters (Northland), with their influx coinciding with El Niño Southern Oscillation (ENSO) La Niña periods when SSTs were higher than the average for this region of New Zealand.

In temperate regions where SST gradually decreases to around 15 °C during winter, populations can persist year round. For example, at Moreton Bay, Australia (an important warm temperate foraging ground), green turtles actively feed at 15 °C (Read et al. 1996), while further south in New South Wales, green turtles remain active in water as low as 12 °C (pers. obs., C. J. Limpus). These winter temperatures are similar to those experienced in northern New Zealand (Chiswell 1994), with the SST regime characterised by the 20 °C isotherm seasonally expanding and retracting across the North Island. To our knowledge, there are no recent reviews that have examined the distribution and occurrence of green turtles throughout New Zealand waters. The aim of this study, therefore, was to revisit the straggler or occasional visitor hypothesis, by compiling 118 years of sighting and stranding data to determine the spatio-temporal distribution and population structure of green turtles in New Zealand. More specifically, evidence was evaluated for (1) an increase in records of turtles over recent decades, (2) persistent, year-round presence of turtles, (3) variation in turtle records with seasons, coasts (west *vs* east coast of northern New Zealand), and anomalies in sea surface temperature.

2.3 Methods

2.3.1 Sighting, stranding and incidental capture data

Sighting, stranding and incidental capture data from New Zealand between 1895 and 2013 were compiled to form the New Zealand Marine Turtle Sightings and Stranding Database (NZMTSSD) for this study. Compiled data included date, location, record type (sighting, stranding, or incidental capture), and, biometric data (CCL). All records from Gill (1997) were combined with validated records of sightings or strandings collected between 1997 and 2013. Sources included published and unpublished (grey) literature, validated reports from individuals of the public, and records from government and non-government organisations. In this work, “New Zealand” refers to both North and South Islands, encompassing those outlying islands on the continental shelf within the 200 m isobath and the seas out to the 200 nautical mile Economic Exclusion Zone (EEZ). Records from New Zealand’s territorial Kermadec Islands (ca. 900 km northeast of New Zealand at ~30° S) were omitted from this study because this subtropical island group is biogeographically distinct from mainland New Zealand.

Records were initially categorised into three ‘types’: Sighting (sightings of live, free-ranging green turtles), Stranding (green turtles found alive or dead, either as beach cast or floating at sea), or Incidental capture (by-caught green turtles in recreational or commercial fishery activities). For the present analysis, ‘Incidental’ captures were included with all ‘Sightings’ data, as both record ‘types’ were considered to represent live, free-ranging turtles at the time of capture.

To determine variation in the mean number of records through time (within each quarter (season)-year) R software (R Development Core Team 2014) was used to fit a generalised linear model (McCullagh and Nelder 1989), assuming a log-link function and a Poisson error distribution. Counts of records were modelled using the predictors T: Type (factor with two levels: Sighting/Incidental capture and Stranding), C: Coast (factor with two levels: East and West), Y: Year (continuous), Q: Quarter (factor with four levels: Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec), QTA: Quarterly Temperature Anomaly (continuous), and ENSO: El Niño Southern Oscillation Multivariate Index (continuous). The most complex model involved fitting three-way interactions between Type, Coast, and each of the other four predictors, i.e. Y×T×C, Q×T×C, ENSO×T×C, and QTA×T×C. A backwards stepwise procedure based on the AIC criterion was subsequently applied to remove extraneous predictors. Consequently, all terms involving ENSO were removed from the model. Many two-way interactions and two three-way interactions (namely C×T×Y and C×T×Q) were retained in the model. Eight records from prior to 1983 were removed from this analysis because they were too temporally sparse to be informative.

Some specific hypotheses regarding interaction terms were tested in post-hoc analyses using the *phia* package in R (De Rosario-Martinez 2015). Where Year interacted with other terms in the model, a significant effect of Year within and across the levels of the interacting term was tested. Specifically, a significant effect of Year was tested within each combination of Coast and Type, while a difference in the Year-slope between Coasts within each Type was also tested. Interactions between Quarter (season) and Type and Coast were also investigated.

QTA was calculated using a 47 year time series of the SST taken at University of Auckland’s Leigh Marine Laboratory (c. 36.3° S, 174.8° E) on the east coast north of Auckland. Averages were calculated for each quarter in each year, and then the anomaly was taken by subtracting the respective mean for each quarter across all years, giving the deviation from the expected long-term average for that quarter. Although the Leigh coastal SST dataset was collected from a single location, the data are accordant with the range, amplitude, and variability described for northern New Zealand by Chiswell (1994) and Duffy (2002). Therefore the Leigh coastal SST dataset was chosen as a suitable broad-scale regional proxy for the average long-term inshore SST regime for both coasts of northern North Island.

To investigate seasonal variations in sighting and stranding patterns between the west and east coasts of New Zealand, a two-dimensional Pearson's χ^2 test was applied. A one-way analysis of variance (ANOVA) was used to compare season (Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec) and the size of stranded turtles (CCL) to investigate the influence of SST (season) upon the stranding of different size classes. Distribution maps were created using ESRI ArcMap version 10.1.

2.3.2 Size-class, sex ratio and maturity

Standard CCL measurements (±0.1 cm) were recorded for all turtles recovered during the course of this study (Limpus et al. 1994, Bolten 1999). Biometric data were then augmented with CCL data obtained from the New Zealand Department of Conservation Herpetofauna Database, marine turtle rehabilitation records from Kelly Tarlton's Sea Life Aquarium (Auckland), and Gill (1997). To examine the green turtle population structure CCL measurements were grouped into 5 cm size class categories and the frequency distribution described (Limpus et al. 1994, Seminoff et al. 2002b, Arthur and Balazs 2008). In addition, 'new' and 'recent' recruits were identified following the descriptions of Limpus et al. (2005).

Where dead stranded turtles were recovered during the course of the study, gross examinations were undertaken, with sex and maturity status determined by *in situ* examination of the gonads and associated ducts (following criteria of Rainey (1981), and Limpus and Reed (1985)). Maturity status was categorised as immature or mature (Limpus et al. 2005, Meylan et al. 2011). Sex ratio was calculated as the proportion of identifiable females in the sample. To test the null hypothesis that there was no significant departure from a sex ratio of 2:1 within the samples (the demonstrated sex ratio for green turtles within eastern Australia (Limpus et al. 2005, Limpus 2008a), a χ^2 test with Yates correction factor was applied.

2.4 Results

2.4.1 Green turtle spatial distribution in New Zealand

In total, 194 green turtle records were obtained, comprising 36 previously published records (Gill, 1997, encompassing records from 1985 to 1996), and 158 unpublished records (encompassing records from 1997 to 2013). Records were primarily from the North Island (north of ca. 41° S); only 3 records were reported from the South Island, with the most southerly record from Birdlings Flat, Canterbury (ca. 43.8° S). Ninety-two percent (n = 178) of all records were from Auckland northward (ca. 37° S) (Figure 1). Stranded turtles comprised 65% (n = 126) of records, while sightings and incidentally captured turtles comprised 25% (n = 49) and 10% (n = 20), respectively. A large proportion of records (69% of those between 1895 and 2013) were from the east coast of the North Island. More specifically, nearly all sightings of free-ranging turtles (98%, n = 48) and almost two thirds of all strandings (66%, n = 83) occurred on the east coast of the North Island. There was only one reported sighting of a free-ranging turtle from the west coast. Of the 20 documented incidental captures, seven occurred on the west coast and thirteen occurred on the east coast. All records (except two) were recorded within New Zealand’s inshore waters (0-200m). The two records of green turtles offshore were from north-east of New Zealand; a sighting approximately 24 km west of North Cape (at ~570 m water depth) and an incidental capture by a commercial shallow long-line fishing vessel approximately 300 km west of Whangarei (at ca. 35° S and ca. 2,000 m water depth). The incidentally captured turtle was a juvenile estimated at < 45 cm CCL and clearly exhibited pronounced plastron ridges, white ventral surface and sharp carapace edges characteristic of an oceanic phase individual.

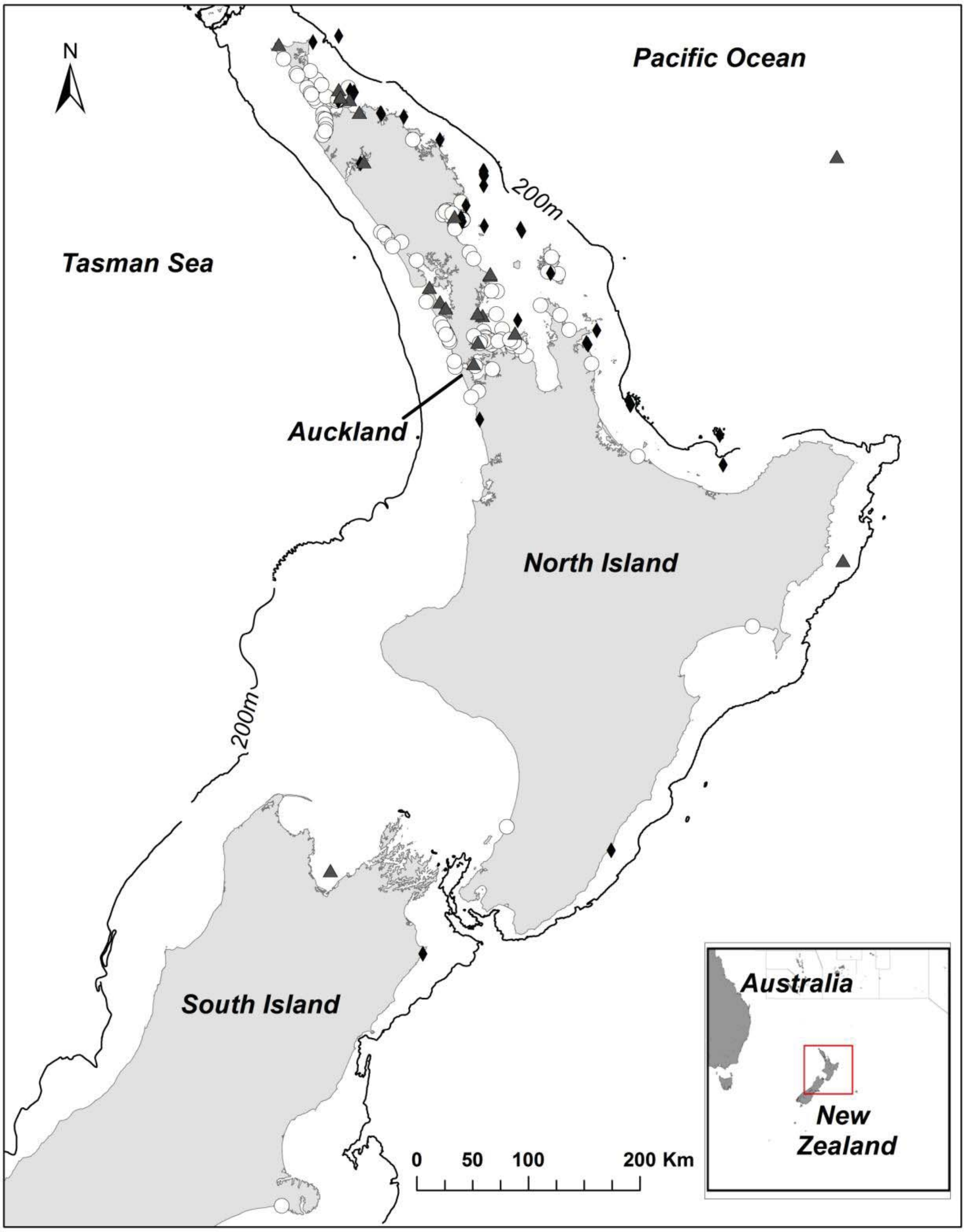

Figure 1. Distribution of green turtle records from New Zealand from 1895 to July 2013 (n = 194). Stranded turtles (○); Sightings (♦); Incidental captures (▲).

2.4.2 Size-class, sex ratio and maturity

The size-class frequency distribution of all measured turtles (CCL) encompassed a range of 17.6–94.6 cm (μ = 51.9 cm, S.D. = 11.6, *n* = 86) (Figure 2). The smallest turtle recorded was considered an oceanic-phase turtle based on size (17.6 cm CCL) and morphological characteristics. The next smallest turtle had a CCL of 37.3 cm. No green turtles in the hatchling size range were observed. Of 64 turtles examined externally, 8% (*n* = 5) of turtles exhibited morphological characteristics of 'new' recruits (μ = 40.8 cm CCL, S.D. = 3.3, range 37.3-44.8 cm), while a further 39% (*n* = 25) turtles were defined as 'recent' recruits (μ = 45.6 cm CCL, S.D. = 2.5, range 41.9-52.5 cm); new and recent recruits comprised 47% (μ = 44.8 cm CCL, S.D. = 3.2, range 37.3-52.5 cm, *n* = 30) of all turtles examined externally. New or recent recruits were encountered every month of the year, suggesting no temporal pattern of settlement.

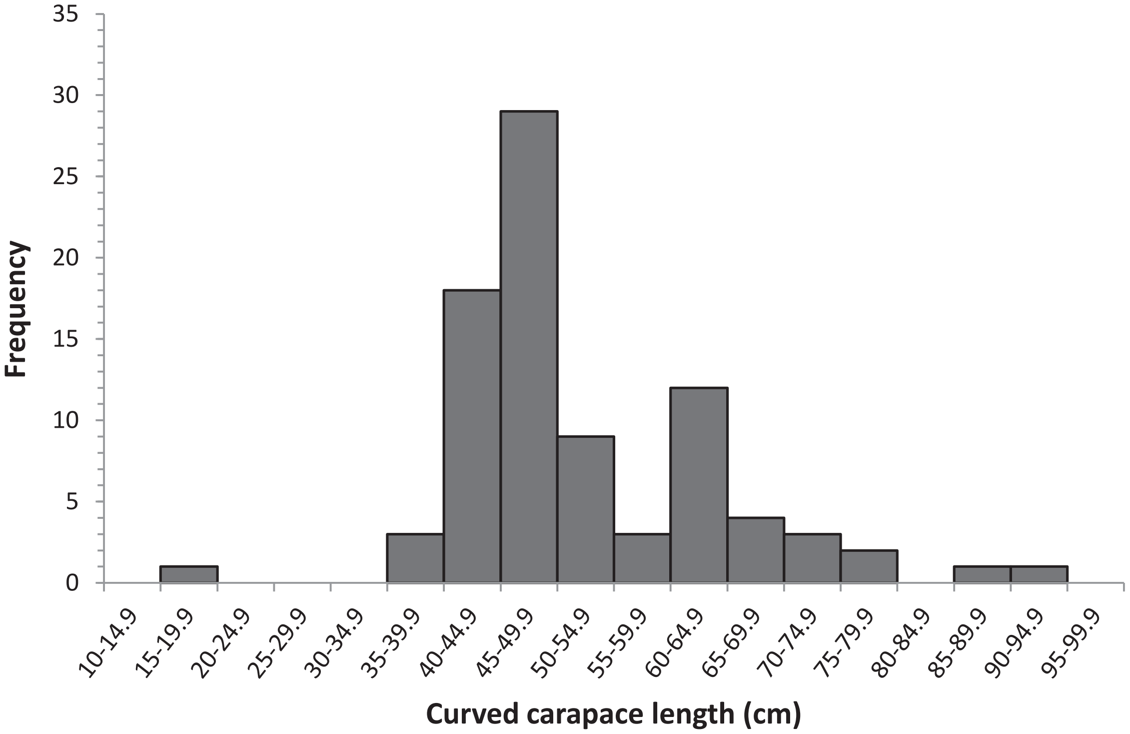

Figure 2. Size class frequency distribution of green turtles in New Zealand recorded from 1895 to 2013. Mean Curved Carapace Length (CCL) = 51.9 cm (S.D. ± 11.6 cm, *n* = 86).

Of 41 green turtles examined, 15 were males, 25 were females, and one undetermined. All turtles examined were immature juveniles. In all males, the testes were undeveloped and inactive, while in all females, the ovaries were inactive, with unexpanded stroma with no sign of vitellogenesis. Expressed as a F:M sex ratio (1.7:1), there was no significant difference from 2:1 ($\chi^2 = 0.3$, $p = 0.58$, df = 1 with Yates correction factor). A *t*-test revealed no significant difference in the size (CCL) between male and female turtles ($t = 0.97$, $p = 0.34$).

2.4.3 Seasonal patterns

The generalised linear model (GLM) selected by the AIC contained the predictors Type, Coast, Year, Quarter, QTA, and a number of interactions (Table 1). Quarter (season) effects were different among combinations of Type and Coast (Table 1). This three-way interaction was not significant at the 5% level, however, the AIC model selection process favoured retaining it in the model. Quarter had a significant effect on east coast sightings ($\chi^2 = 12.00$, $p = 0.0296$; Figure 3c) with July-September (winter) being significantly more likely to have low sightings. However, there was no significant effect of Quarter on strandings on the west coast ($\chi^2 = 8.58$, $p = 0.1064$; Figure 3a) and east coast ($\chi^2 = 3.32$, $p = 0.6899$; Figure 3b), and sightings on the west coast ($\chi^2 = 0.90$, $p = 0.8255$), though it is likely that these tests were low powered. Two-way interactions showed that, overall (across coasts), quarterly effects were not significant for either Sightings ($\chi^2 = 5.78$, $p = 0.2457$) or Strandings ($\chi^2 = 5.25$, $p = 0.2457$). However, Quarter did show a strongly significant effect on the east coast overall ($\chi^2 = 14.07$, $p = 0.0056$), with records significantly less likely to occur during July-September (winter). This effect corresponds to the pattern of sightings described above. Quarter did not significantly affect west coast records ($\chi^2 = 2.03$, $p = 0.5654$). The one-way ANOVA showed no significant differences in the CCL of stranded turtles and season within which they stranded ($F_{3,72} = 1.141$, $P = 0.338$, Table 2).

Table 1. Analysis of deviance table for a Poisson generalised linear model of the number of green turtle records.

Predictor	LR Chisq	df*	Pr(>Chisq)
QTA[1]	4.265	1	0.0389
Coast	46.221	1	0.0000
Type	16.418	1	0.0001
Year	106.986	1	0.0000
Qtr[2]	5.646	3	0.1302
QTA×Coast	5.852	1	0.0156
Coast×Type	7.079	1	0.0078
Coast×Year	0.01	1	0.9214
Type×Year	0.329	1	0.5665
Coast×Qtr	12.881	3	0.0049

[1] QTA: Quarterly Temperature Anomaly

[2] QTR: Quarter (season)

*: degrees of freedom

Table 2. One-way ANOVA comparing Curved Carapace Length (CCL) of stranded turtles and the seasons in which they were recorded (Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec; *n* = 76).

Source o variation	SS[¥]	df*	MS[♦]	F	*P*-value	*F* crit.
Between Groups	465.708	3	155.23	1.141	0.3384	2.731
Within Groups	9796.206	72	136.05			
Total	10261.91	75				

[¥] SS: sum of squares

[♦] MS: mean squares

*: degrees of freedom

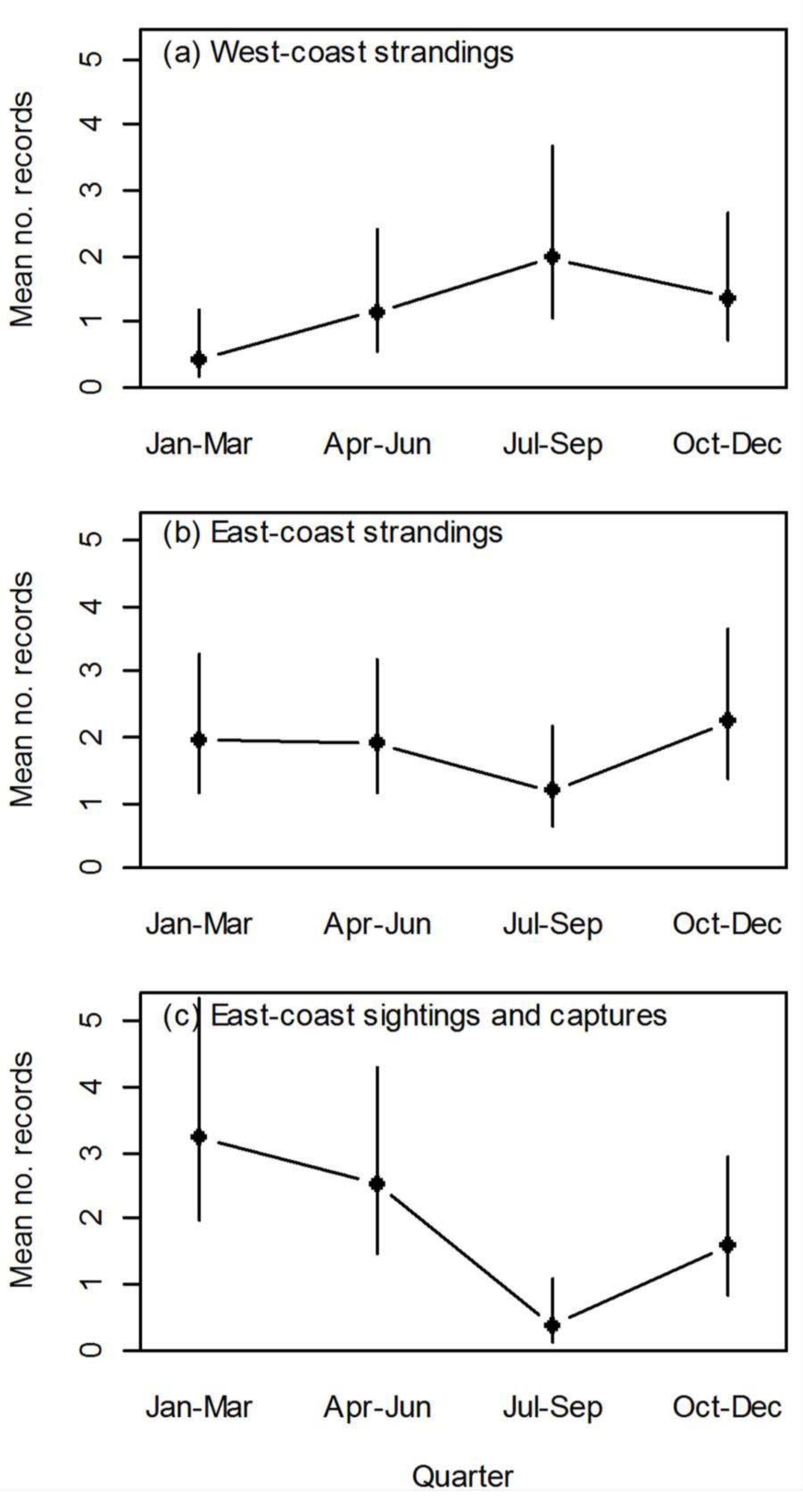

Figure 3. Predicted means from a Poisson generalised linear model of the number of sightings and strandings per quarter for 2014, assuming typical temperatures. Error bars represent 95% confidence intervals around the means. A, West-coast strandings; B, East-coast strandings; C, East-coast sightings and captures.

2.4.4 Sea surface temperature

The coastal SST regime for northern North Island calculated from the Leigh dataset had a range average of 14 °C (August) to 20.6 °C (February), giving an amplitude of 6.6 °C for the North Island's northeast. The lowest recorded temperature at Leigh was 12.3 °C and the highest was 23.9 °C, with an average annual long-term mean of 17.2 °C. There was a significant negative effect of Quarterly Temperature Anomaly ($\chi^2 = 4.27$, $p = 0.0389$, Table 1) on the number of records overall, suggesting there were fewer records when it was unusually warm. The QTA effect interacted with the predictor Coast ($\chi^2 = 5.85$, $p = 0.0156$) and, when tested within Coasts, the result was significant for the west ($\chi^2 = 9.42$, $p = 0.0043$) but not the east coast ($\chi^2 = 0.40$, $p = 0.5271$). This suggests that there were more strandings on the west coast when there were unusually low temperatures recorded at Leigh. There was no evidence of more strandings on the east coast during unusually low sea surface temperatures; however, strandings were predicted to be more consistent throughout the year (Figure 3b). In addition, there was no evidence of a difference in the effect of QTA between strandings and sightings; therefore, the QTA:Type interaction was excluded by the model. There was also no detectable relationship between the El Niño Southern Oscillation (ENSO) Multivariate Index and records of turtles.

2.4.5 Annual trends

The number of green turtle reports has increased each decade since records began, with 98% of all records reported since 1983. From 1983, green turtles were reported every year, with the greatest number recorded during 2011 (18) and 2012 (25). Overall, there were more strandings than sightings (Likelihood ratio $\chi^2 = 16.4$, $P = 0.0001$, Table 1). There was a significant overall linear increase in the log-mean number of records over the 30 years of this study ($\chi^2 = 107.0$, $p < 0.0001$, Table 1). This corresponds to an exponential increase on the untransformed scale (Figure 4). Overall, there was no significant difference between the Year-slopes of the east and west coasts for strandings ($\chi^2 = 1.49$, $p =$

0.2223). However, the rates of increase in sightings over time were marginally different between the east and west coasts ($\chi^2 = 5.06$, $p = 0.0496$).

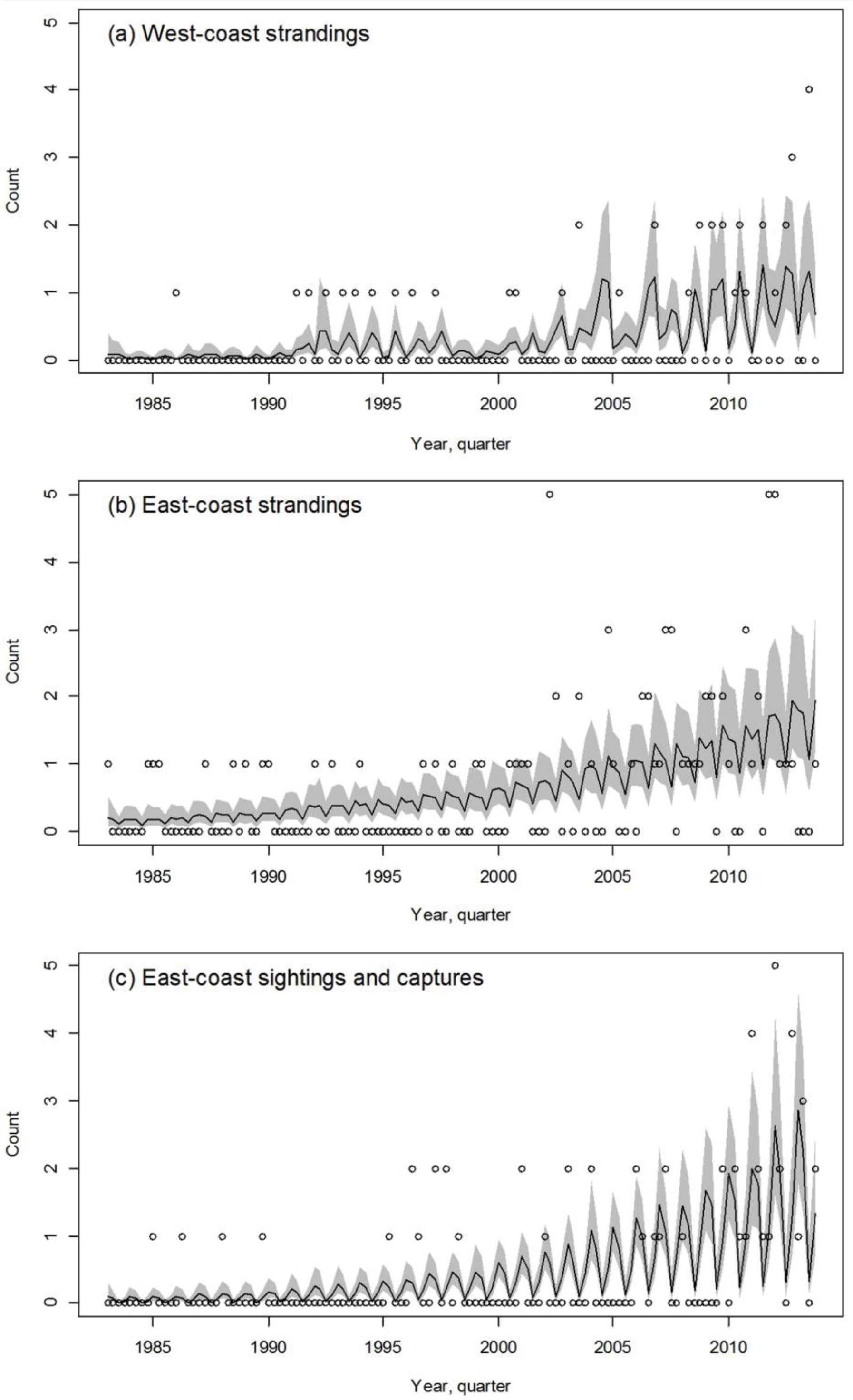

Figure 4. Predictions from a Poisson generalised linear model of the mean of the quarterly counts of turtle records between 1983 and 2013. Circles represent the number of counts recorded, the black lines are the predicted means, and the grey areas show the 95% confidence bounds for the means. **A**, West-coast strandings; **B**, East-coast strandings; **C**, East-coast sightings and captures.

2.5 Discussion

2.5.1 Distribution

Sighting, stranding and incidental capture data reveal a concentration of green turtle occurrence across northern New Zealand; a region of New Zealand's landmass that dissects the western boundary of the South Pacific sub-tropical gyre system (SPSG). The SPSG and its associated currents in the southwest Pacific (Southern Equatorial Current (SEC), East Australian Current (EAC), and Tasman Front (TF)) have been described as the key transport mechanism for post-hatchling marine turtles dispersing from southwest Pacific rookeries (Boyle 2006, Boyle et al. 2009). Therefore, it seems likely that oceanic-phase juvenile green turtles passively entrained in the SPSG would eventually arrive in New Zealand via the TF and the sub-tropical East Auckland Current (EAUC) – a natural extension of the TF within the SPSG system.

The influence of the TF and EAUC upon the influx of marine reptiles was initially suggested by Gill (1997). Data presented herein support this, with 69% of the total records of green turtles originating from the northeast inshore region of the North Island. This northeast region of New Zealand is where the EAUC (and associated eddies) are the dominant oceanographic transport system (Roemmich and Sutton 1998, Stanton and Sutton 2003). The TF and EUAC have also been identified as the primary mechanism responsible for the influx of tropical fish species to New Zealand (Francis and Evans 1993, Francis et al. 1999).

2.5.2 Size-class, sex ratio and maturity

Data presented here support the hypothesis that oceanic-phase juvenile green turtles are settling into nearshore neritic habitats following a natural dispersal and settlement pattern. Size-class frequency and sexual maturity data show the New Zealand population comprises a discrete assemblage of post-settlement immature turtles ranging from small juveniles to large sub-adults (μ = 51.9 cm CCL, S.D. = 11.6, range 17.6–94.6 cm, *n* = 86). Size at recruitment and a female biased proportional sex ratio of 0.63 (F:M sex ratio of 1.7:1) within this study are similar to the expected sex ratio for eastern Australian foraging populations (Limpus and Walter 1980, Limpus and Reed 1985, Limpus et al. 1994). The identification of 5 new recruits in the range of 37.3-44.8 cm CCL indicates settlement occurs at ca. 40.8 cm CCL.

Only 35% of examined turtles were new or recent recruits, with the remaining 65% (*n* = 34), comprised of older age classes up to large sub-adults (μ 63 cm CCL, S.D. = 9.9, range 46.5–94.6 cm). If turtles settle at ca. 40.8 cm CCL as shown, this suggests these older age classes have occupied coastal neritic habitats for some time, hence exhibiting a degree of residency. The absence of mature adult turtles does suggest however, if residency does occur, that turtles are departing from New Zealand coastal areas prior to the onset of puberty. Discrete assemblages of immature age classes at neritic foraging grounds and puberty related migrations have been observed in other regions (Koch et al. 2007, Meylan et al. 2011).

Immature-dominated cohorts occupying neritic foraging areas were initially identified by Carr and Caldwell (1956) and later defined by the term *developmental migrations* (Carr et al. 1978). This concept denotes the post-settlement progression of juveniles transitioning through a series of developmental habitats as they change in size and ecological requirements prior to ultimately settling at an adult foraging ground (Carr 1980, Musick and Limpus 1997, Bolten 2003, Moncada et al. 2006, Meylan et al. 2011). In these studies, size-class partitioning was influenced by differing habitat requirements between age classes, e.g., food availability, predation risk, and dispersal and migratory behaviour (Moncada et al. 2006, Koch et al. 2007, Bresette et al. 2010, Meylan et al. 2011). Such patterns in size class partitioning may then be occurring in New Zealand waters, where northern neritic habitats constitute a temporary developmental habitat, with juvenile green turtles dispersing across the southwestern Pacific transitioning through this habitat for a period of time.

2.5.3 Seasonality

Further evidence supporting a temporary developmental habitat in northern New Zealand was the consistent year round occurrence of green turtles shown in the present work. Contrasting Gill (1997), this work found no correlation between ENSO (La Niña periods) and the influx of green turtles, and, despite an average winter SST of 14 °C, sightings of live free-ranging turtles were recorded in all seasons. This contradicts with the currently accepted view that green turtles appear incidentally as stragglers or occasional visitors (McCann 1966, Gill 1997). Nevertheless, data presented here detected a significant decrease in sightings on the east coast during winter, despite stranding rates remaining relatively stable throughout the year. Several possibilities may lead to lower detection probability during colder seasons: 1) a seasonal reduction in marine observers (e.g. recreational fishers and divers) during winter (Gill 1997), 2) a compensatory response to cooler water by temporarily migrating to warmer water (offshore) during winter (Mendonca 1983, Epperly et al. 1995, Ogren and McVea 1995), 3) turtles employing a ‘sit and wait’ behaviour during winter, where turtles undergo long quiescent dives of up to several hours with very short surface intervals (Hochscheid et al. 2005). Despite this, the present work does indicate a more permanent occurrence of green turtles in New Zealand’s neritic habitats than previously concluded (Gill 1997). The temperature regime described by the Leigh dataset does suggest overwintering turtles would experience some degree of temperature related stress, however, their occurrence in similarly cool winter temperatures has been observed in Australia. For example, at Moreton Bay, Australia (a permanent warm temperate foraging ground), green turtles have been shown to actively feed at 15 °C (Read et al. 1996), while in New South Wales, green turtles remain active in water as cool as 12 °C (pers. obs., C. J. Limpus).

Although season did not affect the occurrence of green turtles overall, strandings on the west coast were significantly affected by QTA. On the west coast, there was an increase in strandings during winter, and this was found to be significantly higher during unusually cold winters (reflected by the QTA at Leigh). A correlation between westerlies and southerlies, and colder SST at Leigh has been observed in other studies (Greig et al. 1988). Therefore, it is suggested that the prevalence of stronger westerly winds and higher storm frequency during winter may substantially impact green turtle populations within the west coast, resulting in higher incidence of stranded turtles within this region. Correspondingly, a marginal decrease in stranded turtles on the east coast during winter may reflect an inverse effect, where stronger offshore westerly winds reduce the probability of stranding incidence. Similarly, a correlation between onshore storm events and turtle strandings has been observed on the east Australian coast (Boyle 2006). One unexpected result was the negative effect of the QTA where fewer records were observed when the SST (based on the QTA) was unusually warm. However, the negative effect of the QTA could be attributed to the fact that the majority of records were of stranded animals, hence, when SST are unusually warm, fewer temperature-related strandings will result in fewer stranding numbers overall during these periods. Also, no relationship between CCL and season was detected; suggesting environmental conditions are exerting equal influence across the population.

2.5.4 Annual trends

The strandings and sightings data show an exponential increase in the number of green turtle records over the time period examined. This increase in records is attributed at least in part to an increase in the number of observers reporting sightings and strandings as a result of public awareness efforts during the course of this study. Similarly, increased observer presence (e.g. SCUBA divers, game fishers, and commercial fisheries) over the last 30 years has resulted in increased numbers of newly recorded tropical and subtropical fishes from north-eastern North Island (Francis et al. 1999). Nevertheless, other factors that may influence the observed increase should not be discounted. For example, the observed increase may parallel conservation efforts in the region. In the southwestern Pacific, the nearest potential source populations are those at the Southern Great Barrier Reef off Australia’s east coast. These populations have recovered over the last several decades following significant declines due to human over exploitation and other anthropogenic mortality (Limpus 2008). Consequently, if New Zealand green turtles do derive from these populations, it would expected the increase in abundance of oceanic-phase juveniles dispersing through the region as a result, would eventually be reflected in the number of observations in New Zealand waters.

Increasing evidence shows that climate change is leading to rapid poleward shifts in the distributional range of marine species (Sorte et al. 2010, Burrows et al. 2011, Feary et al. 2013). These climate-mediated extensions have been observed in a range of tropical fish populations, though most substantially throughout the east coast of Australia, driven by an associated strengthening of the subtropical gyre and poleward penetration of the EAC over the last 60 years (Booth et al. 2007, Ridgway 2007, Last et al. 2011). The strengthening of the EAC has resulted not only in the broadening and extension of this boundary current and its associated eddy system but also led to an increase in the SST of the Tasman Sea (Cai et al. 2005, Neuheimer et al. 2011). Hence, such changes in oceanographic conditions could theoretically transport more juvenile green turtles from eastern Australian rookeries (or farther afield) toward New Zealand waters. Once they disperse to this region of the south Pacific, they would remain in this oceanic developmental habitat until they settle into neritic habitats in northern New Zealand at the size observed here in this study.

2.5.5 Conclusions

Empirical evidence presented here supports the hypothesis that green turtles occurring in New Zealand neritic habitats reflect a natural post-oceanic settlement behaviour rather than oceanic-phase stragglers incidentally blown ashore by storm and other stochastic events. It is proposed that New Zealand comprises a temperate intermediary habitat where oceanic-phase green turtles dispersing across the southwest Pacific settle and transition through this habitat for a period of time before departing to other regions. This contrasts with the prevailing view that green turtles represent stragglers incidentally blown off course or occasional visitors that seasonally inhabit New Zealand’s inshore waters. Although this study has provided some important insights into our understanding of green turtles in New Zealand, several questions still remain. Investigating residency and foraging behaviour (Chapter 4) would provide further insight into temporal occupancy and habitat use. Furthermore, identifying the genetic origin of the population (Chapter 3) will assist with establishing region wide population connectivity. This is particularly important for defining dispersal patterns and migratory pathways. Additionally, for green turtles along the edge of their range, monitoring their response to temperature shifts in relation to rising global temperatures (as noted in Pilcher et al. 2014) will be of fundamental importance for managing this species in the southwest Pacific.

Chapter 3 **Connectivity across the Pacific: origins of green turtles (*Chelonia mydas*) foraging in New Zealand waters**

3.1 Abstract

Effective conservation management of threatened marine turtle species requires an understanding of population dynamics including range, migratory pathways and the identification and extent of foraging grounds utilised. Post-settlement immature green turtles (*Chelonia mydas*) occupy coastal habitats year round in New Zealand, constituting the most southern foraging aggregation identified in the Pacific Ocean. To date, no study has investigated the natal origins of turtles in this foraging aggregation or their interrelated dispersal or migratory pathways. To characterize the genetic structure and elucidate regional connectivity of turtles at this foraging ground, genetic analysis of ~770 base pair sequence of mitochondrial (mt) DNA was conducted on 42 stranded immature green turtles. Fifteen haplotypes, including one orphan haplotype were identified. The unique aggregation comprises turtles carrying haplotypes from green turtle management units in the western Pacific and endemic eastern Pacific clade. This mixed stock foraging aggregation represents a distributional overlap of 14 recognised Management Units (MU) originating from widely dispersed rookeries throughout the Pacific Ocean. Thus, results presented underscore wide ranging population connectivity across the region which has notable management implications for the respective MUs. This study also provides the first genetic assay of eastern Pacific green turtles in the southwestern Pacific region, substantially extending the distributional range of this species. Data presented reveals trans-oceanic dispersal routes into the southwestern Pacific not previously identified for eastern Pacific rookeries. An observed significant difference in size class of two *C. mydas* cohorts reflect different post-hatchling dispersal scenarios, developmental migrations and oceanic residency times for juvenile green turtles in the south Pacific.

3.2 Introduction

Effective conservation of threatened species requires the identification of discrete population boundaries and congruent conservation territories that encompass the entire spatio-temporal range occupied during their life cycle (Campbell and Godfrey 1994). However, delineating populations for conservation become challenging for wildlife managers when species exhibit cryptic life histories which make them inherently difficult to study. These challenges become particularly apparent when attempting to manage wide ranging marine species such as sharks (Castro et al. 2007, Jorgensen et al. 2010), cetaceans (Baker et al. 1999, Andersen et al. 2001, Witteveen et al. 2004), and migratory fish (Pecoraro et al. 2016) that often occupy an array of spatio-temporally disparate habitats throughout their lives. For instance, while conservation efforts of a threatened species may be successful in one region, adverse effects in another geopolitically remote region may go undetected, hence continue to drive population decline overall (Polidoro et al. 2011). This scenario is further exacerbated in philopatric species because they can be demographically autonomous and therefore unlikely to be replenished by immigration following decline (Avise 1998). Thus, it is important to determine stock resolution and regional connectivity for species of conservation concern to ensure appropriate scales of jurisdiction and conservation are achieved.

With the advent of molecular analyses techniques in recent decades, researchers have been able to gain important insights into threatened species' more cryptic biological aspects such as: population structuring (Andersen et al. 2001); biogeographical distribution of genetic lineages and populations (Portnoy et al. 2014); inter-population spatial relationships (Grant et al. 1980); and migratory behaviour (Palsboll 1999, Castro et al. 2007, Pecoraro et al. 2016). Using maternally inherited mitochondrial (mt) DNA markers, researchers are able to identify population specific genomic sequences to investigate aspects of molecular evolution, phylogeography, and population demography (Moritz et al. 1987, Avise and Bowen 1994, Norman et al. 1994). This fundamental understanding of population biology has been critical in underpinning conservation efforts in a vast number of threatened species (Avise 1998). For example, mtDNA analyses has identified regionally distinct stocks of the threatened white shark (*Carcharodon carcharias*) in the northern and southern hemispheres (Pardini et al. 2001, Jorgensen et al. 2010), and at a finer spatial scale discrete population structuring in the Australian white shark population (Blower et al. 2012). Consequently, the eastern and southwestern Australian sub-populations are now considered as independent management units for conservation (DSEWPaC 2013). Where wide ranging species have been heavily exploited in the past, genomics has been pivotal in identifying discrete breeding stocks, migratory routes, and mixed-stock feeding grounds that have been able to inform trans-boundary conservation and management strategies (Hoelzel 1994). For instance, humpback whales (*Megaptera novaeangliae*) are a migratory species that were hunted to near extinction during the past two hundred years until commercial whaling was banned in 1966 (Baker and Clapham 2004, Fleming and Jackson 2011). Subsequently, molecular genetics has been used to identify regional breeding stocks, migratory corridors, and feeding grounds, which in turn have facilitated international and regionally coordinated recovery strategies for this species worldwide (Baker and Clapham 2004, Witteveen et al. 2004, Olavarría et al. 2007, IWC 2010, Fleming and Jackson 2011, Carvalho et al. 2014).

As highly migratory species, marine turtle populations typically encompass a diverse array of breeding and nesting areas, migratory pathways and foraging grounds that all need consideration to ensure species protection occurs across their entire spatio-temporal range. Because marine turtles exhibit complex life history traits (Bolten 2003, Godley et al. 2010), defining stock boundaries remains challenging. In addition, given that marine turtles disperse and migrate vast distances across international jurisdiction (Bolten et al. 1998, Vargas et al. 2015), the conservation of these threatened species often require trans-boundary coordination (Lewison et al. 2004, Blumenthal et al. 2009). For the globally endangered green turtle (*Chelonia mydas*), genetic studies have resolved several key factors including global population structure, matriarchal phylogeny, female-mediated gene flow, and the behavioural aspects of natal homing (Carr 1975, Meylan et al. 1990, Bowen et al. 1992, Encalada et al. 1996). Across the Pacific

Ocean, the distribution and structure of green turtle populations have been shaped by evolutionary, behavioural and climatic processes over the past ~3 million years (Bowen et al. 1992, Dethmers et al. 2006, Duttonet al. 2014). Today, a mosaic of genetically distinct breeding stocks (management units: MU) that are inter-dispersed at mixed stock foraging aggregations span the entire region (Sterling et al. 2013, Dutton et al. 2014, FitzSimmons and Limpus 2014, Naro-Maciel et al. 2014, Read et al. 2014, Jensen, Bell, et al. 2016, Jensen, Pilcher, et al. 2016).

Despite considerable advances in our understanding of green turtle biology, large areas of the Pacific remain understudied, particularly in reference to neritic foraging aggregations (Sterling et al. 2013) and the pelagic phase of post-hatchling and juvenile green turtles (FitzSimmons and Limpus 2014). Accordingly, identifying and monitoring populations at foraging grounds of mixed origin stock has become a prominently critical facet of conservation management for green turtles at a regional level (Encalada et al. 1996, Abreu-Grobois et al. 2006). An added advantage of connecting source rookeries with often distant foraging and developmental grounds, is that it provides an opportunity to identify critical dispersal and migratory pathways together with insights into the oceanographic mechanisms that influence them (Luschi et al. 2003, Blumenthal et al. 2009, Godley et al. 2010). With an understanding of their spatial ecology, conservation managers can identify key areas where human activities overlap and potentially negatively impact marine turtle species (Godley et al. 2010). For example, mtDNA analysis revealed that post-hatchling loggerhead turtles (*Caretta caretta*) originating from southwest Pacific rookeries were being incidentally captured as juveniles in the east Pacific Peruvian longline fishery (Boyle et al. 2009). This research not only provided insight into dispersal mechanisms and developmental migrations of marine turtles in the south Pacific, it also highlighted the need for regional cooperation between nation states to reduce mortality at sea.

In New Zealand, green turtles occupy neritic habitats across the North Island (Gill 1997, Godoy et al. 2016). Recent research suggests this foraging aggregation constitutes a transitional developmental habitat for post-settlement immature juveniles to large sub-adults and delimits the southernmost foraging aggregation in the Pacific Ocean (Godoy et al. 2016; Chapter 4). As yet, no study has investigated the natal origins or genetic structure of turtles within this aggregation. Given New Zealand's proximity to Australia and the largest breeding rookeries in the Pacific Ocean (Limpus 2008a), it's plausible to expect that post-hatchling turtles dispersing from eastern Australian rookeries via the East Australian Current (EAC) and Tasman Front (TF) would be transported southeastward toward New Zealand (Boyle 2006, Godoy et al. 2016). However, although it is highly likely that the aggregation comprises post-settlement individuals from the southwest Pacific Ocean, Dethmers et al. (2010) found that proximate distance or size of the breeding population were not good predictors of the proportional contribution a source stock can have on a foraging aggregation. Therefore, given the wide dispersal characteristics of green turtles, contributions from other rookeries in the Pacific region cannot be discounted. Consequently, this study aims to: 1. identify the genetic origin and describe the composition of the green turtle foraging aggregation in New Zealand, 2. Compare genetic structure within New Zealand to other regional foraging grounds, 3. provide insight into regional connectivity in the context of dispersal and migratory pathways. Findings will be discussed in relation to conservation implications for green turtles across the region.

3.3 Methods

3.3.1 Sample collection

In conjunction with a demographic study of green turtles in New Zealand, 42 skin samples were collected from live and dead stranded turtles between 2006 and 2014. The study aggregation comprises a biogeographically discrete assemblage of neritic foraging immature to sub-adult turtles across northern New Zealand, ca. 34.5°-38.5° S (Godoy et al. 2016; Chapter 4). Small skin samples (~1.0 cm^2) were biopsied from the upper shoulder area of live turtles using a sterile scalpel blade (MUAEC protocols 10/05, 11/62, 14/73). Skin samples were collected from the same location from dead turtles either during necropsy or *in situ* at the stranding site if carcasses were not recovered. Tissue samples were preserved in 70-95%

EtOH. For each turtle, curved carapace length (CCL) measurements (±0.1 cm) were taken (Limpus et al. 1994, Bolten 1999).

3.3.2 DNA extraction and amplification

Genomic DNA was extracted from tissue samples (~0.1g) using proteinase K digestion in 300 µl of extraction buffer containing 40 mM Tris, 20 mM EDTA-Na_2, 100 mM NaCl and 1.5% sodium dodecyl suphate (SDS) (as per Jensen et al. 2013). DNA was recovered from solution by EtOH precipitation in the presence of 3.75 M ammonium acetate and resuspended in TE buffer (10 mM Tris, 1 mM EDTA-Na_2, pH 7.5). A portion of the mtDNA control region (~800 bp) was amplified via polymerase chain reaction (PCR) using the primers LtSeaT (5' -GCATTGGTCTTGTAAACCAAAG-3'), which was modified from LTEi9 (Read et al. 2015), and H950 (5' -AGTCTCGGATTTAGGGGTTTG-3') (Abreu-Grobois et al. 2006). All PCR amplifications included a negative (template free) control reaction to test for contamination (Read et al. 2015). PCR products were dispatched to Macrogen Inc. (Korea) for forward and reverse sequencing.

3.3.3 Data analysis

Sequence chromatograms were aligned manually using the software package Geneious 6.1.8 (Biomaters; available from www.geneious.com). Control region haplotypes were identified using the Basic Logical Alignment Search Tool (BLAST) to compare aligned sample sequences with the online registry of known green turtle reference haplotypes at GenBank (http://www.ncbi.nlm.nih.gov/genbank/) and the US National Marine Fisheries Service (Southwest Fisheries Science Centre) (Jensen et al. unpubl. data). All assigned haplotypes were aligned and truncated to 770 bp for statistical analyses, and haplotype naming convention followed those described in GenBank (http://www.ncbi.nlm.nih.gov). Published haplotype data from 14 Pacific breeding stocks and eight Pacific foraging grounds were collated for statistical comparison to haplotype composition within the New Zealand foraging aggregation (Figure 1).

Genetically distinct breeding stocks (MUs) consisted of: 1. southwestern Pacific: southern Great Barrier Reef (sGBR), northern Great Barrier Reef (nGBR), Coral Sea (CS: comprising Coral Sea platform and Chesterfield Islands, New Caledonia), New Caledonia (d'Entrecasteaux Islands); 2. western central Pacific: Palau, Micronesia, Guam/CNMI, Marshall Islands; 3. south central Pacific: Vanuatu, American Samoa; 4. eastern Pacific: Revillagigedo Islands (Mexico), Michoacan (Mexico), Costa Rica (Guanacaste), Galapagos Islands (Ecuador) (Dutton et al. 2008, Dutton et al. 2014, Dutton et al. 2014, Read et al. 2015, Jensen, Bell, et al. 2016).

Regional foraging grounds consisted of: 1. eastern Australia: Moreton Bay (MB); Shoalwater Bay (SB); Edgecombe Bay (EB), Howick Group (HG), Clack Reef (CR), Torres Strait (TR); 2. New Caledonia: Grand Sud Lagon (GLS); 3. Hawaiian archipelago (HW) (Dutton et al. 2008, Read et al. 2015, Jensen, Bell, et al. 2016). Rookeries (breeding stocks) that did not share haplotypes with the New Zealand sample (e.g. Hawaii and French Polynesia) were excluded from the analyses. Arlequin ver. 3.5.2.2 (Excoffier and Lischer 2005) was used to estimate haplotype diversity (*h*), nucleotide diversity (π), compute conventional pairwise F_{ST} for population comparisons and exact tests of population differentiation. Alpha was set at 0.001 and significance values were obtained from 10,000 permutations. A minimum spanning haplotype network was constructed in PopART (Bandelt et al. 1999) to illustrate the relationship between subject green turtle rookeries and the New Zealand foraging aggregation based on 770 bp mtDNA sequence data. A two-tailed *t*-test was used to compare the mean CCL of turtles with haplotypes found in different regions of the Pacific.

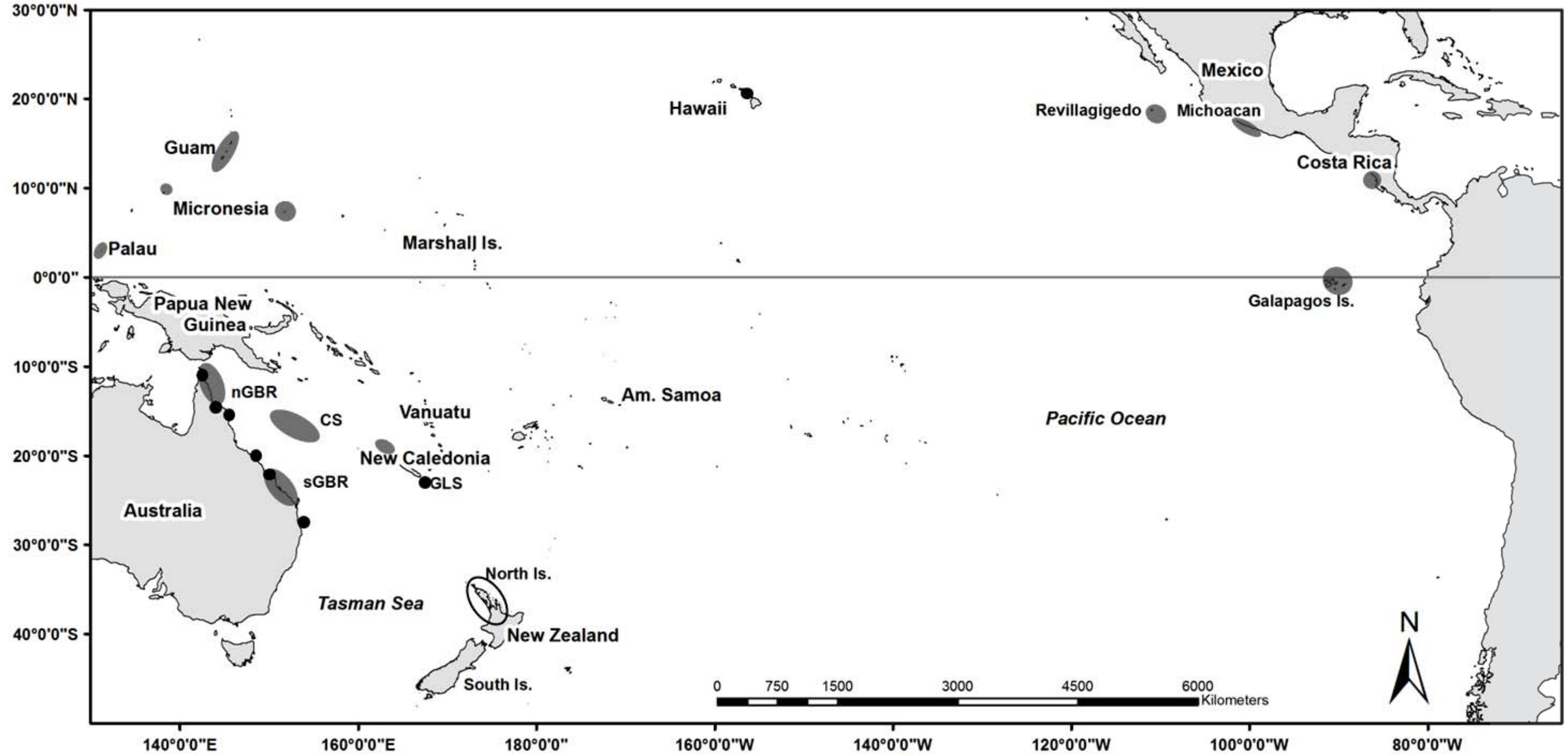

Figure 1. Location of green turtle rookeries in the southwest Pacific (sGBR: southern Great Barrier Reef; nGBR: northern Great Barrier Reef; CS: Coral Sea; New Caledonia; Vanuatu); western central Pacific (Palau; Micronesia; Guam/CNMI; Marshall Islands); south central Pacific (American Samoa); eastern Pacific (Revillagigedo Islands and Michoacan, Mexico; Costa Rica; Galapagos Islands, Ecuador). Points depict locations of foraging grounds in eastern Australia (Moreton Bay; Shoalwater Bay; Edgecombe Bay, Howick Group, Clack Reef, Torres Strait), Grand Sud Lagon (GLS), New Caledonia, and the Hawaiian Archipelago. The New Zealand study site is encircled. Map created with ESRI ArcMap v10.1.

3.4 Results

Except for a small stranded oceanic-phase turtle (17.6 cm CCL), all sampled turtles were immature and sub-adult green turtles (μ = 52.2 cm CCL, SD = 13.2, range = 17.6-94.6 cm), and derive from a neritic foraging aggregation that occupies New Zealand's northern waters, between ca. 38°–34° S, year round (Godoy et al. 2016 Chapter 1). A total of 15 haplotypes were identified with 63 polymorphic sites, comprising 53 substitutions and 10 indels. Fourteen haplotypes have been previously identified at rookeries in the Pacific region, while one orphan haplotype (CmP80.4) from a single individual was identified and accessioned in GenBank ID: KX685265 (Table S1: Appendix 2). This previously undescribed haplotype differed from CmP80.1 in a single nucleotide substitution at position 157.

Three haplotypes found in high relative frequency across southwest Pacific rookeries (sGBR, Coral Sea, New Caledonia) comprised 52% of the haplotypes found in the New Zealand aggregation i.e. CmP47.1 (43%, *n* = 18), CmP80.1 (7%, *n* = 3) and CmP85.1 (2%, *n* = 1) (Table S1: Appendix 2). All new (*n* = 3) and recent recruits (*n* = 10) identified morphologically (μ = 44.1 cm CCL, SD = 3.6, range = 37.3-49.5 cm) carried these same haplotypes: CmP47.1 (*n* = 9), CmP80.1 (*n* = 3) and CmP85.1 (*n* = 1). In addition, the small oceanic-phase stranded green turtle also carried the haplotype CmP47.1. Haplotypes reported from rookeries of the west central Pacific (Marshall Islands, Micronesia, Palau, Guam/CNMI) and south central Pacific (American Samoa) contributed only 7% to the New Zealand aggregation (CmP22.1: 5%, *n* = 2 and CmP20.1: 2%, *n* = 1). In contrast, haplotypes endemic to east Pacific rookeries (Revillagigedo, Michoacan, Costa Rica, Galapagos Islands) comprised 29% in total i.e. CmP4.6 (10%, *n* = 4), CmP4.7 (10%, *n* = 4), CmP4.1 (5%, *n* = 2), CmP24.1 (2%, *n* = 1), CmP4.4 (2%, *n* = 1) and CmP4.9 (2%, *n* = 1). The minimum spanning network (Figure 2) illustrates the broadly regional grouping of related breeding stocks in the west, central, and southwest Pacific compared to the east Pacific; and the proportional composition shared with the New Zealand aggregation.

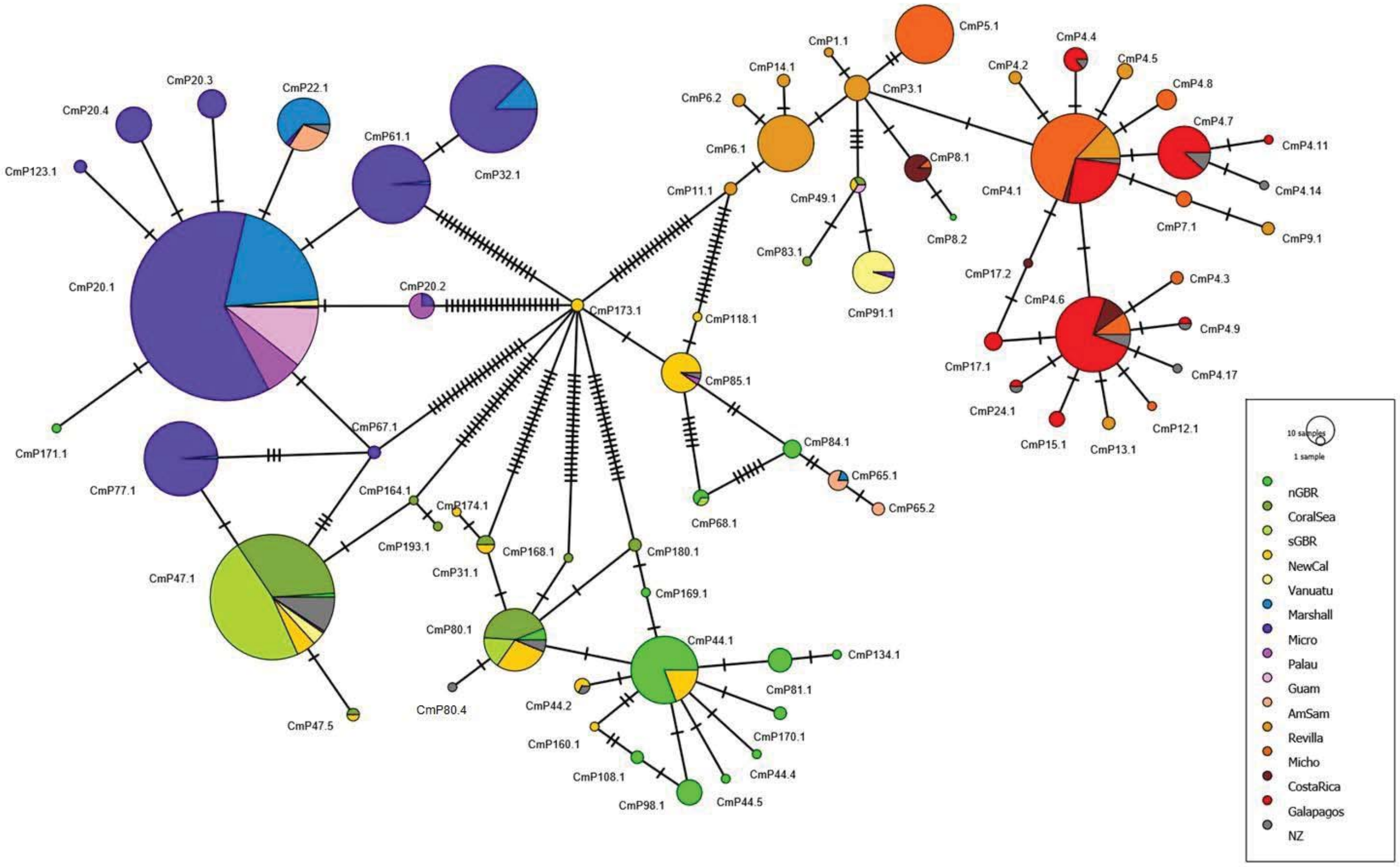

Figure 2. Minimum spanning haplotype network showing the relationship of Pacific Ocean green turtle rookeries and the New Zealand foraging aggregation based on 770 bp mtDNA. Each hatch mark represents a single base pair difference between haplotypes.

The ten turtles with endemic east Pacific haplotypes combined with two other turtles assigned the closely related haplotypes: CmP4.14 and CmP4.17 (Figure 2), exhibited morphological characteristics typical of green turtles encountered in the eastern Pacific (Pritchard and Mortimer 1999, Parker et al. 2011), while all other turtles sampled exhibited morphological traits typifying central and west Pacific regions (Pritchard and Mortimer 1999, Limpus 2008a; Figure 3). Excluding the single oceanic-phase green turtle considered an outlier (17.6 cm CCL), a *t*-test revealed a very highly significant difference in the size (CCL) between turtles with endemic east Pacific haplotypes and others with haplotypes found across central, western, and southwestern Pacific breeding stocks ($t = 7.50$, df = 37, $P < 0.001$). Green turtles with east Pacific haplotypes were significantly larger ($\mu = 65.5$ cm CCL, SD = 10.2, range = 53.4-94.6 cm, $n = 14$) than their counterparts ($\mu = 46.1$ cm CCL, SD = 6.0, range = 37.3-65.3 cm, $n = 25$; Figure 4).

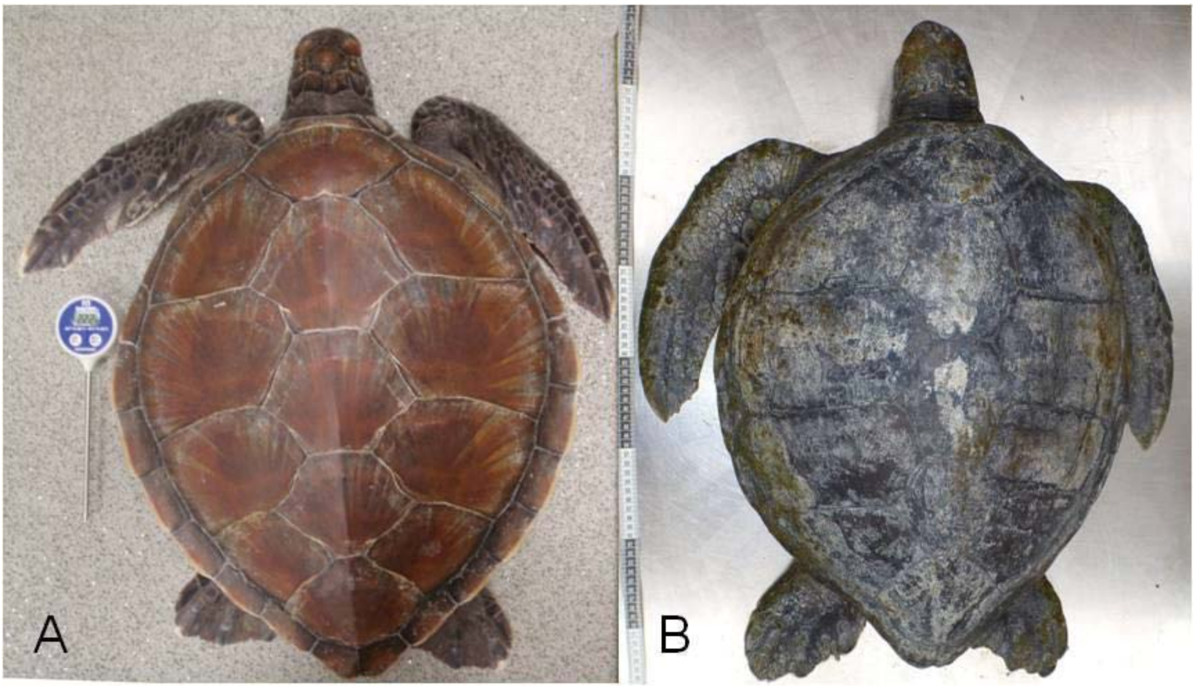

Figure 3. Immature green turtles typically encountered in New Zealand waters exhibiting morphological characteristics associated with A, west and central Pacific rookeries; B, east Pacific rookeries.

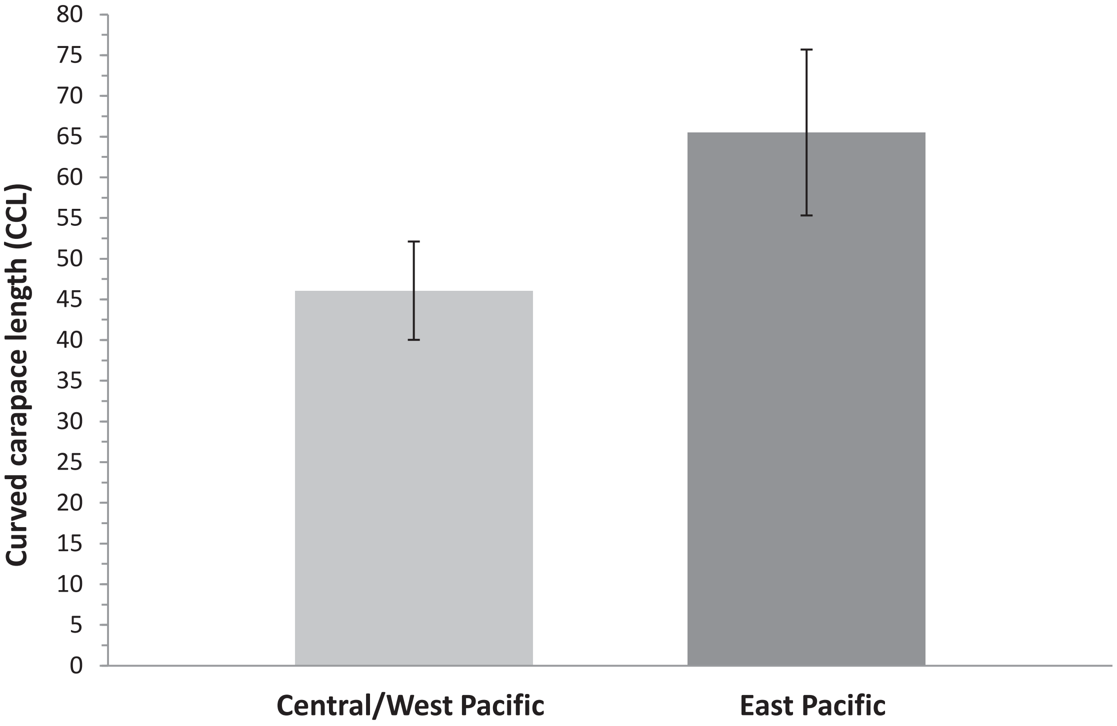

Figure 4. Mean curved carapace length (CCL ± S.D.) of green turtles from the New Zealand aggregation with West/Central Pacific haplotypes (μ = 46.1 cm CCL, SD = 6.0, n = 25) and East Pacific haplotypes (μ = 65.5 cm CCL, SD = 10.2, n = 14).

Estimates of genetic diversity (Table 1) were higher in the New Zealand foraging aggregation (h = 0.8026, π = 0.0271) than the nearby Australian foraging grounds of Moreton Bay (h = 0.3570, π = 0.0106), Shoalwater Bay (h = 0.3303, π = 0.0094) and Edgecombe Bay (h = 0.4178, π = 0.0132), yet were similar to foraging grounds further north i.e. Howicks Group (h = 0.8272, π = 0.0229), Clack Reef (h = 0.8303, π = 0.0205) and Torres Strait (h = 0.7519, π = 0.0182). Similarly high diversity indices to the New Zealand aggregation were also observed at the foraging ground Grand Lagon Sud, New Caledonia (h = 0.8520, π = 0.0271). Pairwise F_{st} and exact tests show very highly significant haplotype differentiation (P<0.001) between the New Zealand foraging aggregation and all Pacific breeding stocks except the Coral Sea MU which was highly significant (F_{st} = 0.0740, P<0.01; Table 2). Similarly, haplotype differentiation between the New Zealand foraging aggregation and all other regional foraging grounds was very highly significant (P<0.001), except for Grand Lagon Sud, New Caledonia which was highly significant (F_{st} = 0.0408, P<0.01).

Table 1. *Chelonia mydas* mtDNA control region sequence diversity across Pacific breeding stocks and regional foraging grounds illustrated by haplotype diversity (*h*) and nucleotide diversity (*π*) ± standard deviation (SD). Sample size (*n*) and number of haplotypes (*H*) for each sample location are shown.

		n	*H*	*h* ± SD	*π* ± SD
	New Zealand	42	15	0.8026 ± 0.0581	0.0271 ± 0.0135
Breeding Stock	sGBR	102	3	0.6614 ± 0.0568	0.0083 ± 0.0044
	nGBR	81	14	0.1640 ± 0.0473	0.0063 ± 0.0034
	Coral Sea	97	10	0.4938 ± 0.0515	0.0164 ± 0.0083
	New Caledonia	64	13	0.8323 ± 0.0238	0.0247 ± 0.0123
	Vanuatu	31	5	0.5204 ± 0.0961	0.0221 ± 0.0112
	Marshall Islands	128	6	0.4596 ± 0.0467	0.0018 ± 0.0012
	Micronesia	538	11	0.6765 ± 0.0169	0.0055 ± 0.0030
	Palau	36	3	0.3317 ± 0.0879	0.0027 ± 0.0017
	Guam/CNMI	48	2	0.0417 ± 0.0395	0.0021 ± 0.0014
	American Samoa	17	4	0.6176 ± 0.1063	0.0220 ± 0.0116
	Revillagigedo	77	11	0.6941 ± 0.0499	0.0018 ± 0.0013
	Michoacan	120	8	0.6371 ± 0.0285	0.0023 ± 0.0015
	Costa Rica	20	6	0.8105 ± 0.0551	0.0034 ± 0.0021
	Galapagos	126	10	0.7346 ± 0.0234	0.0018 ± 0.0013
Foraging Ground	Moreton Bay	125	14	0.3570 ± 0.0552	0.0106 ± 0.0055
	Shoalwater Bay	178	16	0.3303 ± 0.0450	0.0094 ± 0.0049
	Edgecombe Bay	169	13	0.4178 ± 0.0449	0.0132 ± 0.0067
	Howicks Group	161	24	0.8272 ± 0.0187	0.0229 ± 0.0113
	Clack Reef	91	19	0.8303 ± 0.0258	0.0205 ± 0.0102
	Torres Strait	263	35	0.7519 ± 0.0260	0.0182 ± 0.0090
	Grand Lagon Sud	164	19	0.8520 ± 0.0152	0.0271 ± 0.0133
	Hawaii	788	6	0.4637 ± 0.0181	0.0018 ± 0.0012

Table 2. Pairwise F_{ST} values (based on conventional haplotype frequencies) and pairwise exact test *p*-values between New Zealand foraging ground and Pacific breeding stocks and regional foraging grounds. Pairwise F_{ST} significant *P* values shown as * (<0.05), ** (<0.01), and *** (<0.001). Distance between New Zealand foraging ground and compared sites are shown.

			New Zealand	
		Minimum straight distance (km)	F_{ST}	exact test
Breeding Stock	sGBR	2,500	0.2638***	0.0000***
	nGBR	3,800	0.2649***	0.0000***
	Coral Sea	2,600	0.0740**	0.0000***
	New Caledonia	2,200	0.1076***	0.0000***
	Vanuatu	2,400	0.3197***	0.0000***
	Marshall Islands	4,800	0.3894***	0.0000***
	Micronesia	5,000	0.2669***	0.0000***
	Palau	6,300	0.4130***	0.0000***
	Guam/CNMI	6,300	0.5838***	0.0000***
	American Samoa	2,800	0.2372***	0.0000***
	Revillagigedo	9,900	0.2501***	0.0000***
	Michoacan	10,500	0.2712***	0.0000***
	Costa Rica	11,600	0.1619***	0.0000***
	Galapagos	10,400	0.1743***	0.0000***
Foraging Ground	Moreton Bay	2,200	0.1391***	0.0000***
	Shoalwater Bay	2,700	0.1640***	0.0000***
	Edgecombe Bay	3,000	0.1100***	0.0000***
	Howicks Group	3,700	0.0526***	0.0000***
	Clack Reef	3,700	0.0851***	0.0000***
	Torres Strait	4,300	0.1843***	0.0000***
	Grand Lagon Sud	1,600	0.0408**	0.0000***
	Hawaii	9,600	0.4354***	0.0000***

3.5 Discussion

3.5.1 Genetic structure

The New Zealand foraging aggregation shared a large proportion of haplotypes commonly found in rookeries of the southwestern Pacific i.e. sGBR, Coral Sea and New Caledonia. Although these similarities were expected given their proximity to New Zealand, results showed no clear relationship with a single genetic stock; instead highlighting the contribution from several genetically distinct management units. Although mixed stock foraging grounds are frequently observed in marine turtle feeding aggregations globally (Bass et al. 2006, e.g. Bowen and Karl 2007, Blumenthal et al. 2009, Dethmers et al. 2010), what makes the New Zealand assemblage unique is that it shares haplotypes with widely dispersed stocks from across the Pacific Ocean region, and in particular, a large proportion of individuals carrying haplotypes from very distant east Pacific rookeries located > 9,000 km. In addition, due to the presence of east Pacific green turtles in the New Zealand aggregation, it does not resemble the genetic structure of any other Pacific foraging ground examined in this study. Although, east Pacific morphotypes have been infrequently observed in foraging grounds of eastern Australia (Limpus et al. 2005, Jensen, Bell, et al. 2016), none have been identified from genetic studies on foraging grounds in the southwestern Pacific (Dethmers et al. 2010, Read et al. 2015, Jensen, Bell, et al. 2016). Thus, this work provides the first empirical evidence of a genetic link to east Pacific stocks in the southwestern Pacific. Consequently, the unique genetic structure at this temperate foraging ground reveals previously undefined regional connectivity, and provides evidence for trans-oceanic dispersal for east Pacific green turtles into the southwestern Pacific.

3.5.2 Oceanic dispersal and migratory pathways

Ocean currents play an important role in the dispersal and aggregation of marine turtles (Carr 1980, Bolten 2003, Luschi et al. 2003, Bass et al. 2006, Dethmers et al. 2010). In the Pacific Ocean, the equatorial currents associated with the North and South Pacific gyre systems disperse post-hatchlings from rookeries across the tropical and sub-tropical Pacific (Boyle et al. 2009, Naro-Maciel et al. 2014). In the South Pacific Ocean, hatchling dispersal from nesting rookeries, as well as the dispersal of juveniles and adults, is influenced by oceanic features associated with the anti-cyclonic South Pacific subtropical gyre (Limpus et al. 1994, Boyle and Limpus 2008, Boyle et al. 2009, Pendoley and Christian 2012). For example, electronic, satellite, and flipper tag data collected from juvenile and breeding adults reveal a generalised westward migration and dispersal pattern which is consistent with the direction of the South Equatorial Current (SEC) (Trevor 2010, Pendoley and Christian 2012). Marine turtles dispersing or migrating via the SEC may eventually enter the Coral Sea and be carried southward down the East Australian Current (EAC), and potentially eastward into the Tasman Front (TF). Pendoley & Christian (2012) for example, reported a juvenile green turtle that had been initially tagged and released at Scilly Atoll, Tahiti (ca. 16° S 154° W), was eventually found 17 months later, ca. 4500 km west at Norfolk Island (ca. 29° S 168° E). Concordantly, hatchlings emerging from eastern Australian rookeries are considered to passively disperse via the EAC into the western boundary of the South Pacific Gyre (Boyle and Limpus 2008). It is in this anti-cyclonic gyre system and associated eddies, where it is believed they may reside for several years in oceanic habitats before they return and settle into neritic foraging grounds of eastern Australia at approximately 30–45 cm CCL (Limpus et al. 2005, Jensen, Bell, et al. 2016).

Stranding data of post-hatchling green turtles from eastern Australia lends support for this oceanographic mediated process, indicating a southward dispersal from sGBR waters via the East Australian Current, marginally past New South Wales and then out into the Pacific Ocean via the Tasman Front (Limpus et al. 1994, Walker et al. 2006). Given that New Zealand lies at the southwestern extremity of the Polynesian triangle, its landmass dissects the same area of the western boundary of the South Pacific subtropical gyre. Therefore, it’s plausible that green turtles dispersing either initially westward from New Caledonia and the Coral Sea (via SEC associated currents: Read et al. 2015) and then south into the EAC, or, eastward from Australian rookeries

(Boyle 2006), could become passively entrained within the Tasman Front, eventually dispersing toward New Zealand (Chapter 1). Data presented here supports this pattern of dispersal where the majority of the green turtles in this study carried haplotypes that originate from southwest Pacific management units i.e. sGBR, Coral Sea and New Caledonia. Further, all new and recent recruits identified herein, were of southwest Pacific origin and were similar in size at settlement to their counterparts recruiting to neritic developmental foraging habitats in eastern Australia and New Caledonia (Limpus et al. 2005, Read et al. 2015, Godoy et al. 2016).

A pattern of westward dispersal of post-hatchlings and juveniles via the SEC also explains the occurrence of east Pacific haplotypes in the New Zealand aggregation. The substantially longer migration to New Zealand in comparison to turtles emerging from southwestern Pacific rookeries is indicated by the significantly larger size class of turtles originating from eastern Pacific stocks. It is unclear, however, if individuals from this east Pacific cohort have remained in oceanic habitats until reaching New Zealand or have progressively transitioned through series of neritic habitats while migrating across the Pacific. In contrast to the prevailing view that post-pelagic juvenile green turtles exhibit a high degree of site fidelity once they recruit into neritic developmental habitats, ontogenetic habitat shifts during the juvenile developmental stage has been observed in the Atlantic and Pacific Ocean (Meylan et al. 2011). Supporting evidence for green turtles dispersing westward and remaining in the oceanic zone without transitioning through neritic habitats en route to New Zealand, is the absence of east Pacific haplotypes from southwestern Pacific foraging grounds (Dethmers et al. 2010, Read et al. 2015, Jensen, Bell, et al. 2016). Although some east Pacific morphotypes have been observed at eastern Australia foraging grounds, identification was based on morphological characteristics only and records are highly infrequent, with only a few individuals reported despite several decades of monitoring effort (Limpus et al. 2005, Jensen, Bell, et al. 2016). This contrasts considerably to the large proportion of turtles of east pacific origin observed in the New Zealand aggregation.

3.5.3 Conservation management

Understanding dispersal patterns and regional connectivity is a fundamental aspect of conservation management of marine turtles (Godley et al. 2010). For example, identifying migratory pathways and foraging habitats enable wildlife managers to spatially overlay potentially adverse human activities, such as commercial fishing, to reduce bycatch and mitigate these threats more appropriately (Lewison et al. 2014). Accordingly, understanding the distribution, connectivity and genetic structure of discrete green turtle stocks across the Pacific Ocean has been instrumental in identifying threats and developing coordinated conservation strategies (Seminoff et al. 2015). As with other regions of the world, however, there has been an increasing focus on understanding the more cryptic aspects of marine turtle life histories such as juvenile dispersal and population dynamics at mixed stock foraging grounds (Abreu-Grobois et al. 2006). In this context, this study links genetically distinct green turtle management units from across the Pacific region to a temperate foraging aggregation in the southwestern Pacific not previously described. Thus, the information provided herein supports the objectives outlined within regional conservation initiatives such as the Pacific islands marine turtle action plan (SPREP 2012), and Pacific green turtle recovery plans (NMFS and USFWS 1998a, 1998b).

Chapter 4 **The diet of green turtles (*Chelonia mydas*) at a temperate foraging ground of the southwestern Pacific**

4.1 Abstract

Recent research has provided evidence for the year round presence of green turtles (*Chelonia mydas*) in New Zealand's northern inshore waters; however no study has investigated if individuals in this aggregation are foraging in these neritic habitats. Herein, gut contents of 34 stranded and incidentally caught immature and sub-adult green turtles were analysed to investigate the foraging ecology of this globally endangered species at the edge of their range. This chapter provides the first insight into the diet of green turtles in New Zealand; a transitional temperate habitat in the southwestern Pacific Ocean. Data presented demonstrate that post-oceanic phase green turtles recruit and transition to a benthic foraging strategy within New Zealand's inshore waters. Here they consume a variable diet of macroalgae (89.7 % frequency of occurrence: FO_d), seagrass (27.6 % FO_d), mangrove (10.3 FO_d) and macro invertebrates (75.9 % FO_d). No correlation between size (based on curved carapace length) and diet was identified. In contrast, green turtles of all age classes continued to consume animal prey (predominantly benthic gastropods). The prevalence of animal prey in the diet suggests that green turtles recruiting into nearshore developmental habitats in New Zealand do not ontogenetically transition to a strictly herbivorous diet as they age. Although, no seasonal effect on diet composition was evident, the observed mixed diet foraging strategy may be a facultative response to the environmental constraints on maximizing growth and nutrition in a temperate habitat. This study focused on green turtle diet at the edge of their geographical range, extending our knowledge of the southern limit of green turtle foraging in the Pacific region. Results also highlight the importance of algal-dominated reef and seagrass habitats for this endangered species in New Zealand waters.

4.2 Introduction

Diet and nutrition are fundamental aspects of an organism's ecology because they influence development, morphology, physiology, behaviour, life history and evolution (Raubenheimer et al. 2009, Lihoreau et al. 2015). Furthermore, because food availability is a limiting factor in animal populations, it regulates abundance, distribution, survival and reproduction (Monteith et al. 2014, Resano-Mayor et al. 2016). Therefore, by understanding the diet and foraging strategy of threatened species, wildlife managers can support their conservation as it provides insight into their nutritional interaction with their environment (Berger-Tal et al. 2016). These data can also highlight aspects of their spatial ecology that can assist conservation efforts; such as identifying important foraging habitats that overlap with human activities that can be potentially harmful (McInnes et al. 2016).

Green turtles (*Chelonia mydas*) have evolved to successfully occupy tropical, subtropical and warm temperate seas worldwide (Limpus et al. 1994, Hirth 1997). They exhibit complex life history patterns that encompass coastal nesting areas, neritic foraging grounds, oceanic habitats, and migratory pathways (Balazs 1976, Hirth 1997, Lohmann and Lohmann 1998, Bolten 2003, Luschi et al. 2003, Boyle and Limpus 2008). However, despite their wide global distribution and adaptive success, green turtles are now considered a globally endangered species in the Red List of the International Union for Conservation of Nature (IUCN) (Seminoff 2004). Their decline is due to the direct and indirect adverse effects of human activities upon every life stage of this species (Seminoff 2004). To address this decline, conservation managers require an understanding of their biology across their entire spatio-temporal range. Encapsulated within this paradigm, understanding the foraging ecology of endangered species is a critical component of conservation management because diet and nutrition directly influence growth, reproduction and survival (Bjorndal 1997). Studies have shown the quality and type of food eaten during the developmental phase and inter-nesting periods can have a significant effect on the growth of immature green turtles and the reproductive periodicity of mature adults, respectively (Limpus and Nicholls 1988, Balazs and

Chaloupka 2004, Arthur and Balazs 2008, Kubis et al. 2009). Furthermore, it is important to understand how foraging ecology differs across dynamic marine ecosystems of widely dispersed species in order to formulate regionally specific conservation measures.

Throughout their lives, green turtles undergo several ontogenetic habitat shifts that are coupled with changes in their nutritional requirements and dietary strategies (Bjorndal 1997, Musick and Limpus 1997). For instance, as neonate hatchlings leave the nest they disperse into oceanic pelagic habitats where they reside for several years (Hirth 1997, Musick and Limpus 1997). During this period of development, they occupy this niche as epipelagic omnivores, opportunistically foraging on macroplankton in the water column (Balazs et al. 1987, Boyle 2006, Arthur et al. 2008). After several years and at a size of approximately 30–45 cm curved carapace length (CCL), they recruit to inshore developmental habitats, where they may remain for decades until they reach sexual maturity (Balazs 1985, Balazs et al. 1987, Bjorndal 1997, Chaloupka and Limpus 1997, Zug et al. 2002, Seminoff et al. 2002a, Bolten 2003, Limpus et al. 2005, Koch et al. 2007, Arthur et al. 2008, Boyle and Limpus 2008, Senko et al. 2010). It is at this point that post-settlement green turtles recruiting into neritic habitats transition ontogenetically from an epipelagic omnivorous foraging strategy to a strictly benthic herbivorous foraging strategy, based on a diet of seagrass and macroalgae (Bjorndal 1980, Garnett et al. 1985, Forbes 1996, Bolten 2003). Critically, this transition requires a physiological shift in digestive function where a change in the composition and specificity of gut microflora occurs (Bjorndal 1980). This digestive shift enables hindgut fermentation of seagrass and macroalgae to produce volatile fatty acids (VFA) as a primary energy source (Bjorndal 1979). This shift has been considered by some to occur abruptly and irreversibly in order to maximise digestive efficiency (Bjorndal 1979, 1980, Reich et al. 2007). However, although the basis of herbivory is still regarded as fundamental to green turtle biology today, more recent studies across its range show a degree of plasticity in the diet (Seminoff et al. 2002b, Cardona et al. 2009, 2010, Awabdi et al. 2013, González, Botto, et al. 2014).

There are a growing number of studies across regional populations showing evidence that the transition from omnivory to herbivory is not as abrupt, irreversible or definitive as previously suggested (Hatase et al. 2006, Cardona et al. 2009, González et al. 2012, Morais et al. 2014). Increasingly, animal prey has been identified in the diet of neritic green turtles (Seminoff et al. 2002b, Heithaus et al. 2002, Hatase et al. 2006, Amorocho and Reina 2007, 2008, Cardona et al. 2009, 2010, Quiñones et al. 2010, Carrión-Cortez et al. 2010, Lemons et al. 2011, Burkholder et al. 2011, Awabdi et al. 2013, González, Botto, et al. 2014). The degree to which the diet varies is influenced by a complex interaction between spatio-temporal forage and prey availability (Bjorndal 1980, Garnett et al. 1985, Forbes 1996, Brand-Gardner et al. 1999, González et al. 2012), gut microflora composition (Bjorndal 1979, Bjorndal et al. 1991), individual nutritional requirements and foraging strategy (Forbes 1996, Amorocho and Reina 2007, 2008, Vander Zanden et al. 2013), tidal movement (Fuentes et al. 2006), environmental perturbation (Gama et al. 2016), and predation risk (Burkholder et al. 2011, Meylan et al. 2011). In addition, given that green turtles are poikilotherms, they rely on ambient environmental temperature to support their metabolism, including digestion. As a consequence, environmental temperature will influence dietary selection and consumption rates (Bjorndal 1980, Mendonca 1983, Balazs et al. 1987, Amorocho and Reina 2008, Reisser et al. 2013, Morais et al. 2014). In turn, the influence of temperature on dietary selection may be greater for juveniles than adults because adults, with a larger body size, have greater thermal inertia, hence can maintain higher core body temperature which facilitates better digestive efficiency (Bjorndal 1980, Mrosovsky 1980, Spotila and Standora 1985, Spotila et al. 1997). Also, anatomically, larger turtles have proportionally larger intestinal volumes that increase passage time and fermentation (Bjorndal and Bolten 1990, Morais et al. 2014).

In the southwestern Pacific, the neritic green turtle aggregations at Moreton Bay, eastern Australia (27.3° S, 153.3° E), and Norfolk Island (29.0° S, 168.0° E) constitute the southern limit of green turtle foraging (Limpus et al. 1994, Arthur et al. 2008, Pendoley and Christian 2012). However, recent evidence suggests a neritic aggregation of post-pelagic immature juveniles to large sub-adults exists year round in New Zealand's northern waters (ca. 34°-38° S) (Godoy et al. 2016; Chapter 1). The New Zealand aggregation comprises a mixed stock of turtles genetically linked to discrete management units from the western and eastern Pacific Ocean (Chapter 3). However, no study has yet determined whether turtles in this aggregation are foraging in New Zealand's neritic habitats. If so, this finding would support the residency hypothesis posited by Godoy et al (2016) (see Chapter 2) and would extend the southern limit of green turtle foraging in the Pacific region. Herein, to examine for evidence of foraging, gross necropsies were conducted on green turtles found stranded or incidentally caught in fisheries activities in northern New Zealand between 2006 and 2013. Specifically, it was aimed to 1. determine if green turtles are foraging in New Zealand neritic habits; 2. identify and quantitate the major dietary components and investigate whether a size-correlated ontogenic dietary shift from omnivory to herbivory is evident; 3. investigate the influence of other intrinsic (sex, stock origin) or extrinsic (seasonal) factors on major constituents consumed. Ultimately, by investigating the feeding biology of green turtles in New Zealand, a peripheral temperate habitat for marine turtles in the southwest Pacific Ocean, insight is provided into the foraging ecology of this endangered species in the region.

4.3 Methods

Thirty three stranded green turtles and one incidentally captured individual from New Zealand were examined for diet component analysis. Of the 34 turtles examined, 29 had digesta present in the anterior gastrointestinal tract (oesophagus and stomach), while the remaining 5 were empty and therefore excluded from the statistical analysis (Parmenter 1980). Seventeen turtles were alive when recovered (which subsequently died), hence were considered suitable for diet analysis. The remaining 12 turtles found stranded dead were assessed as D2[1] or D3[2]; decomposition codes describing carcass condition as prescribed by Flint et al (2009). Hence, they were considered suitable for diet analysis given that decomposition was not advanced enough to prevent identification of diet items.

During gross necropsy standard curved carapace length measurements (CCL, ±0.1 cm) were recorded using a flexible fibreglass tape laid over the curve of the carapace (Bolten 1999). CCL measurements were grouped into 5 cm size classes (Limpus et al. 1994, Seminoff et al. 2002b, Arthur and Balazs 2008) with frequency distributions and descriptive statistics (mean, SD, range) calculated to illustrate the size (age) class of the sample. Although it was not possible to conduct histopathological health assessments on all carcasses, to provide a broad subjective assessment of health status each turtle was weighed (kg, ±0.1) and a body condition index (BCI) based on the body weight to CCL ratio was calculated (Limpus et al. 2005). Using the equation BCI = (weight (kg)/CCL^3) x 10^4 (Bjorndal et al. 2000, Limpus et al. 2005), BCI values were categorised and allocated a corresponding subjective visual condition following the criteria of Flint et al (2009):

Very good

- BCI > 1.20: rounded plastron, good muscle mass particularly around the forelimb axilla, neck and hind limbs.

Good

- BCI = 1.11 – 1.20: rounded plastron, good muscle mass particularly around the forelimb and neck area.

Average

- BCI = 1.00 – 1.10: less rounded plastron and muscle mass, loss of fat around forelimb, neck and hindlimbs.

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[1] D2 definition from Flint et al (2009): Dead, carcass in good condition - fresh; suitable for pathology or resembling a carcass fresh enough for eating.

[2] D3 definition from Flint et al (2009): Dead, carcass fair - decomposed but organs intact; autolysis noted on gross examination.

Poor

- BCI < 1.00: flat to concave plastron that is easily depressed. Significant loss of muscle mass and fat deposits around forelimbs, neck, tail and hindlimbs. Sunken eye sockets and often emaciated with substantial epibiont load.

Sex and maturity status was determined by *in situ* gross examination of the gonads and associated ducts following Limpus and Reed (1985):

Males

- Immature: Testis flat or cylindrical, epididymis not pendulous.
- Mature: Testis cylindrical, epididymis distinctly enlarged and pendulous.

Females

- Immature: Ovary with unexpanded stroma and follicles < 0.4 cm diameter. Oviduct white, straight, or only slightly convoluted and less than 1.5 cm diameter.
- Mature: Ovary with expanded stroma. Oviduct pink, very convoluted and at least 1.5 cm in diameter. Small vitellogenic to mature follicles (0.4 – 3.0 cm diameter) may be present. Corpora lutea or healed ovarian scars in the ovary and/ or eggs in the oviduct may also be present.

Where possible, sex and maturity were confirmed histologically. In addition, ‘new’ and ‘recent’ recruits were identified based on morphological characteristics following the descriptions of Limpus et al. (2005).

The complete gastrointestinal tract was removed and the entire contents of the anterior gastrointestinal tract (oesophagus and stomach) were collected and rinsed thoroughly through a 0.5 mm fine mesh sieve. Samples were then fixed in a 4% formalin/clean seawater solution (Seminoff et al. 2002b). Diet components were identified to the lowest taxonomic level possible with a binocular dissecting scope (Balazs 1985, Forbes 1999, Arthur and Balazs

2008). Where digesta material could not be identified to species or genus, a higher taxon was used (Garnett et al. 1985, Seminoff et al. 2002b). Ingested synthetic debris (see Godoy *in prep*, Chapter 5), natural debris (i.e., feathers, bark) and ingested substratum particles (i.e., sand, pebbles, and shell fragments) were also removed and identified.

Diet information was quantified as the percentage frequency of occurrence (*FO*) of each identified dietary item (d) calculated as:

$$FO_d = \frac{(Number\ of\ turtles\ in\ which\ diet\ item_d\ was\ observed)}{Total\ number\ of\ turtles} \times 100$$

While the frequency of occurrence provides an insight into the regularity at which different dietary items are being consumed, it does not illustrate the relative importance of each item (Boyle and Limpus 2008). Therefore, to substantiate the relative importance for each *d*, it's volume was measured through water displacement in a graduated cylinder (±1 ml) (Hellawell and Abel 1971, Forbes 1996, Seminoff et al. 2002b, Carrión-Cortez et al. 2010). The water displacement method uses a graduated cylinder containing water, within which the food sample is submerged, and the increase in volume is recorded (Forbes 1999). For reasonable accuracy, the size of the graduated cylinder should be appropriate for the volume of the sample; that is, displacement of a 1 ml sample should not be measured in a 100 ml graduated cylinder (Forbes 1999). To investigate the importance of dietary items for each turtle and for the New Zealand aggregation overall, measured volumes were used to calculate percentage volume of each diet item identified (V_d) for each turtle, and the total volume (V_t) of each dietary item (d) combined for all turtles. In addition, the mean percentage volume (μ_d) was calculated across all turtles (after Carrión-Cortez et al (2010)).

$$V_d = \frac{(volume\ of\ diet\ item\ (d) in\ a\ turtle)}{Total\ dietary\ volume\ in\ the\ turtle} \times 100$$

$$\mu_d = \frac{\sum \mathrm{V}_d}{n}$$

4.3.1 Data analysis

For statistical purposes, any dietary items that represented less than one percent of the contents in a turtle were considered to be trace amounts and therefore excluded from FO_d calculations. The exclusion of trace items was done to reduce the relative importance of incidentally ingested diet items. However, trace amounts were retained when calculating V_d, V_t and μ_d. Dietary items that were >5% V_d in at least one turtle was considered an important diet constituent (after Garnett et al. 1985, Seminoff et al. 2002b). Dietary items that had a FO_d >25% V_d and FO_d >50% V_d were presented to illustrate which items comprised major components of an individual's diet (modified from Carrión-Cortez et al. 2010).

To determine differences in the ingestion of major dietary items in relation to CCL (as a proxy for age) compositional data V_d and V_t were aggregated into four major categories (c: macroalgae, seagrass, mangrove, and animal prey) and univariate analyses was conducted applying a log linear contrast approach (Aitchison 1983). A linear model (logit transform) was fit to the data using R software (R Development Core Team 2014) to compare the V_d c and V_t c using the covariates CCL, austral season (Summer: Jan-Mar, Autumn: Apr-Jun, Winter: Jul-Sep, Spring: Oct-Dec), sex (male, female), and subpopulation (west Pacific, east Pacific). Seasons were based on the average monthly sea surface temperature (SST) cycle across northern New Zealand (Greig et al. 1988, Godoy et al. 2016). Subpopulation was defined as a possible covariate because two genetically distinct cohorts were identified in the focal aggregation (Chapter 3), and were significantly different in CCL ($t = 6.94$, df = 27, $P < 0.05$) i.e. green turtles with east Pacific haplotypes were significantly larger ($\mu = 67.0$ cm CCL, SD = 10.7, range = 53.9-94.6 cm, $n = 11$) than their west Pacific counterparts ($\mu = 45.3$ cm CCL, SD = 6.2, range = 37.3-65.3 cm, n = 18). However, initial analyses found no correlation between compositional data and the covariates sex and subpopulation, therefore these extraneous predictors were removed from the model and the data pooled for final analysis.

The final linear models: mod = lm (V_d *c* ~ CCL cm + Season) and mod = lm (V_t *c* ~ CCL cm + Season) were fit to the data to test several specific null hypotheses: 1. there was no seasonal effect on the ingested total volume (V_t *c*) of the response variables: total diet volume (four major categories combined), animal prey and macroalgae, while accounting for CCL; 2. there was no seasonal effect on the ingested percentage volume (V_d *c*) of the response variables: macroalgae and animal prey, while accounting for CCL; 3. there was no effect of CCL on the ingested percentage volume (V_d *c*) of the response variables: macroalgae and animal prey, while accounting for season. For tests conducted using V_d *c,* only macroalgae and animal prey were used because seagrass and mangrove data were too sparse to be informative.

4.4 Results

All turtles examined were collected from the upper North Island of New Zealand between latitudes 34.6° - 37.7° S (Figure 1). Of 34 individuals for which the anterior gastrointestinal tract was examined, 29 (85%) had digesta present and 5 (15%) were empty. The five turtles with empty anterior digestive tracts (μ = 44.7 cm, SD = 1.0, range 43.7–45.7 cm) were classed as new (2) or recent recruits (3) and were all categorised as having poor body condition (mean BCI = 0.89, range = 0.77 – 0.93). They were found stranded alive between June (autumn) and October (spring) when SSTs were coldest. Excluding turtles with empty tracts, the total sample for dietary component analysis comprised immature juveniles to large sub-adults (μ = 53.6 cm, SD = 13.4, range = 37.3–94.6 cm, *n* = 29; Figure 2). Of these, 15 (52%) were classed as new (3) or recent (12) recruits (μ = 44.1 cm, SD = 3.6, range = 37.3–49.5 cm) and found stranded alive, stranded dead, or incidentally captured throughout the year in all seasons (Godoy et al. 2016). Of those with digesta present, 12 were males, 15 were females and two were undetermined.

Overall, they had an average mean BCI of 1.03 (SD = 0.13, *n* = 29) that ranged between poor and very good (0.68–1.38).

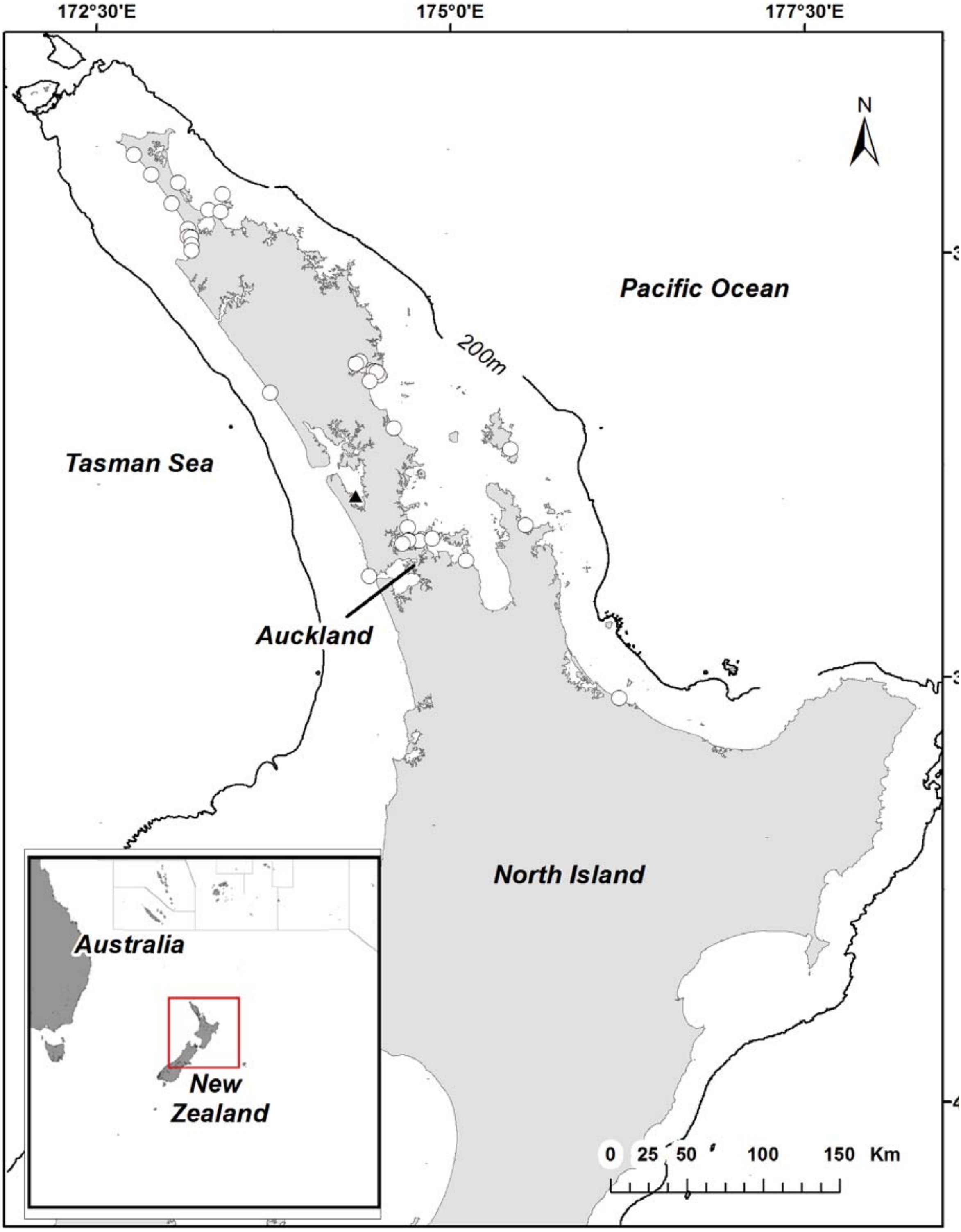

Figure 1. Distribution of stranded (○) and incidentally captured (▲) green turtles collected for diet component analysis from New Zealand.

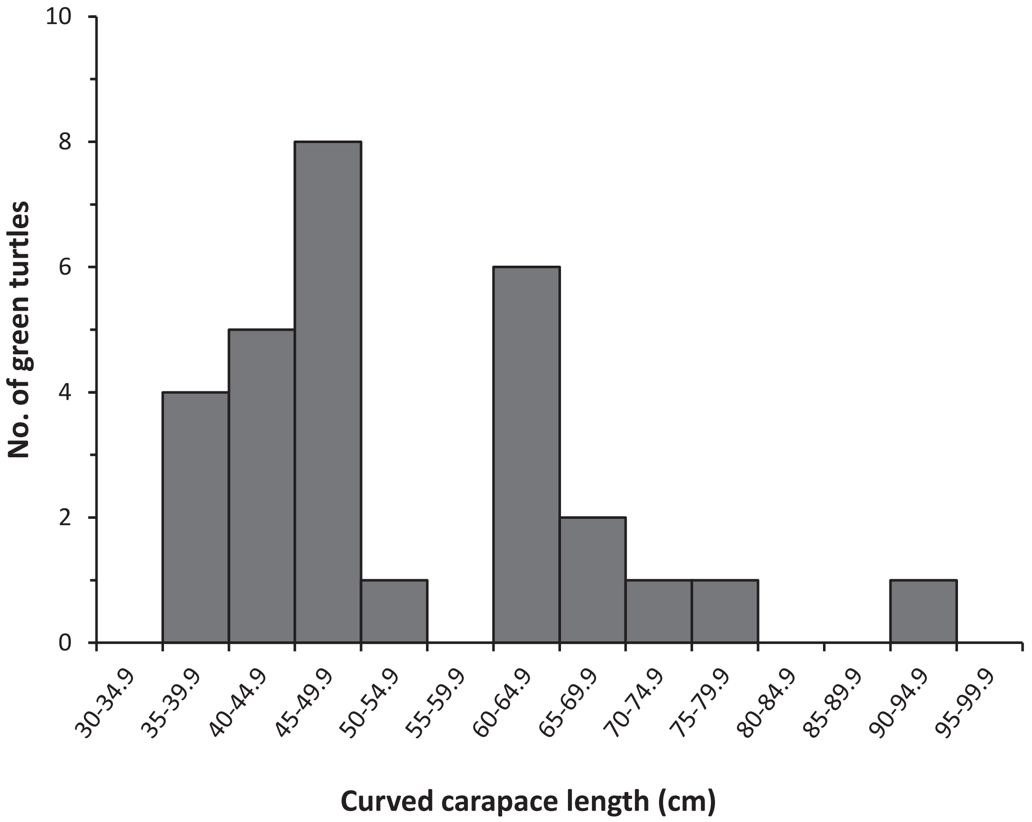

Figure 2. Size class frequency distribution of New Zealand green turtles with digesta present in the anterior gastrointestinal tract (oesophagus and stomach). Mean curved carapace length (CCL) = 53.6 cm (SD ± 13.4 cm, n = 29).

4.4.1 Diet component analysis

A total of 40 taxa were consumed, consisting of macroalgae (n = 27), animal prey (n = 11), one seagrass and one mangrove species (Table 1). Of these, 15 macroalgae species, six animal taxa, the seagrass *Zostera muelleri* subsp. *novozelandica* and the mangrove *Avicennia marina* were identified as important dietary components based on FO_d >5% V_d. A further four macroalgae and four animal taxa were consumed in minor amounts and only by singletons. The remaining nine macroalgae and two animal species were only consumed in trace amounts (<1% V_d). For all turtles combined, macroalgae was the most important diet category consumed in terms of frequency (n = 26 turtles, 89.7 % FO_d) and mean volume percentage (56.7 μ_d). Animal prey also contributed substantially to the diet overall, being frequently consumed (n = 22, 75.9 % FO_d) in large proportions (30.8 μ_d). No pelagic animal prey were identified from the anterior gastrointestinal tract. The seagrass *Z. muelleri* subsp. *novozelandica* ($n = 8$, 27.6 % FO_d, 6.5 μ_d) was moderately represented, while the fruits and leaves of the mangrove *Avicennia marina* ($n = 3$, 10.3 % FO_d, 1.4 μ_d) were observed in comparably lower frequency and volume overall. Natural and synthetic debris accounted for 27.6 FO_d ($n = 8$, 3.8 μ_d) and 10.3 FO_d ($n = 3$, 1.4 μ_d), respectively.

Within the two dominant categories (macroalgae and animal prey), major dietary components were green algae ($n = 19$, 65.5 % FO_d, 26.8 μ_d), red algae ($n = 13$, 44.8 % FO_d, 23.9 μ_d), and benthic gastropods ($n = 9$, 31.0 % FO_d, 11.6 μ_d). Specifically, the green algae *Codium fragile* ($n = 16$, 55.2 % FO_d, 23.8 μ_d), the red algae *Gigatina atropurpurea* ($n = 7$, 24.1 % FO_d, 6.5 μ_d), and the gastropod *Pleurobranchaea maculata* ($n = 5$, 17.2 % FO_d, 4.4 μ_d) were dominant species in the diet. Although some species were not eaten in large quantities by all turtles, in some cases they comprised a large proportion of the gut contents in at least one turtle (i.e. with a FO_d >50% V_d); often appearing as discrete monospecific boluses in the gastrointestinal tract. They include several red algae i.e. *Sarcodia montagneana* ($n = 4$, 3.4 FO_d >50% V_d, 5.3 μ_d), *Pterocladia lucida* ($n = 3$, 3.4 FO_d >50% V_d, 3.3 μ_d), *Gelidium caulacantheum* ($n = 1$, 3.4 FO_d >50% V_d, 2.8 μ_d), *Psilophycus alveatus* ($n = 1$, 3.4 FO_d >50% V_d, 2.7 μ_d); the gastropod *Bursatella leachii* ($n = 2$, 3.4 FO_d >50% V_d, 3.2 μ_d) and egg masses from the neogastropod *Cominella adspersa* ($n = 3$, 3.4 FO_d >50% V_d, 4.0 μ_d).

Table 1. Percentage frequency of occurrence (FO_d) and mean percentage volume (μ_d) of diet components from the anterior gastrointestinal tract (oesophagus and stomach) of green turtles in New Zealand (*n* = 29).

Diet component	*n*	FO_d	FO_d >25 % V_d	FO_d >50 % V_d	μ_d
Macroalgae	26	89.7	69.0	51.7	56.7
Chlorophyta	19	65.5	34.5	20.7	26.8
Codium fragile	16	55.2	31.0	17.2	23.8
Codium convolutum	1	3.4	-	-	0.2
Ulva sp.	3	10.3	3.4	-	2.7
Chaetomorpha aerea	1	3.4	-	-	0.1
Caulerpa flexilis	2	T*	-	-	0.0
Rhodophyta	13	44.8	27.6	24.1	23.9
Gigatina atropurpurea	7	24.1	10.3	3.4	6.5
Sarcodia montagneana	4	13.8	10.3	3.4	5.3
Pterocladia lucida	3	10.3	3.4	3.4	3.3
P. capillacea	1	3.4	-	-	0.7
Gelidium caulacantheum	1	3.4	3.4	3.4	2.8
Psilophycus alveatus	1	3.4	3.4	3.4	2.7
Hypnea sp.	1	3.4	3.4	-	1.0
Acrosymphyton firmum	1	3.4	-	-	0.8
Arthrocardia corymbosa	1	3.4	-	-	0.2
Cladhymenia oblongifolia	1	3.4	-	-	0.4
Sarcothalia sp.	1	3.4	-	-	0.2
Champia novaezelandiae	1	T	-	-	0.0
Gracilaria truncata	1	T	-	-	0.0
Jania rosea	1	T	-	-	0.0
Psaromenia sp.	1	T	-	-	0.0
Ochrophyta	7	24.1	-	-	3.6
Homosira banksii	7	24.1	-	-	2.1
Carpophyllum flexuosum	2	6.9	-	-	1.4
C. plumosum	1	3.4	-	-	0.0
Sargassum scabridum	1	T	-	-	0.0
S. sinclairii	1	T	-	-	0.0
Distromium skottsbergii	1	T	-	-	0.0
Haleopteris sp.	1	T	-	-	0.0

Diet component	n	FO_d	FO_d >25 % V_d	FO_d >50 % V_d	μ_d
Unidentified Algae	10	34.5	-	-	2.5
Seagrass					
Zostera muelleri subsp. novozelandica	8	27.6	6.9	6.9	6.5
Mangrove					
Avicennia marina	3	10.3	3.4	-	1.4
Animal prey	22	75.9	37.9	31.0	30.8
Mollusca	10	34.5	17.2	13.8	13.0
Gastropoda	9	31.0	13.8	13.8	11.6
Bursatella leachii	2	6.9	3.4	3.4	3.2
Pleurobranchaea maculata	5	17.2	6.9	3.4	4.4
Cominella adspersa (egg mass)	3	10.3	3.4	3.4	4.0
Unidentified gastropod eggs	1	3.4	-	-	0.1
Bivalvia	2	6.9	3.4	-	1.4
Musculista senhousia	2	6.9	3.4	-	1.4
Platyhelminthes - Turbellaria	1	3.4	-	-	0.1
Annelida - Polychaeta	1	3.4	-	-	0.1
Arthropoda - Malacostraca	1	3.4	-	-	0.1
Decapoda	1	3.4	-	-	0.1
Isopoda	1	T	-	-	0.0
Cnidaria - Anthozoa	3	10.3	3.4	-	
Actiniaria	1	3.4	3.4	-	1.5
Cnidaria - Hydrozoa	2	T	-	-	0.0
Porifera	5	17.2	-	-	1.3
Unidentified animal prey	11	37.9	17.2	13.8	14.7
Natural substrate and debris	8	27.6	3.4	3.4	3.8
Synthetic debris	3	10.3	-	-	1.4

* T denotes trace levels found

4.4.2 Covariate analysis

Despite a slight reduction in the mean total diet volume (V_t *c*) consumed during winter in comparison to summer, autumn and spring, the linear model showed no significant seasonal effect (taking CCL into account) on the total diet volume consumed ($P = 0.885$; Table 2). Concomitantly, no seasonal effect was observed for the response variables: macroalgae ($P = 0.862$) and animal prey ingested ($P = 0.830$). For percentage volumes (V_d *c*), there was no evidence for a significant effect of season on the ingested percentage volume (V_d *c*) of the response variables: macroalgae ($P = 0.505$; Table 3) and animal prey ($P = 0.693$; Table 3). Equally, while accounting for season, there was no evidence for a significant effect of CCL on the ingested percentage volume (V_d *c*) of the response variables: macroalgae ($P = 0.424$; Table 3) and animal prey ($P = 0.855$; Table 3). Despite no evidence of a seasonal effect on the dietary volumes ingested (V_t *c*, V_d *c*), or a CCL effect on V_d *c*, there was strong evidence of a CCL effect in terms of total diet volume ingested (V_t *c*) i.e. larger turtles consume larger dietary volumes than smaller turtles (Table 2). Between major categories, a negative correlation was observed in the ingestion of macroalgae and animal prey ($r = -0.87$), yet no correlation between any of the other major diet categories was identified.

Table 2. Linear model (logit transform) ANCOVA results comparing the covariates: curved carapace length (CCL) and Season to the response variables: A. Total V_t *c*; B. V_t *c*: macroalgae; C. V_t *c*: animal prey.

A. Total V_t *c*	df ◊	SS ¥	MS ♦	*F*	*P*-value
CCL	1	1919612	1919612	23.51	6.106e-05 ***
Season	3	52729	17576	0.22	0.885
Residuals	24	1959861	81661		
B. V_t *c*: macroalgae					
CCL	1	794826	794826	10.40	0.004 **
Season	3	56887	18962	0.25	0.862
Residuals	24	1833581	76399		
C. V_t *c*: animal prey					
CCL	1	275028	275028	18.07	0.000 ***
Season	3	13411	4470	0.29	0.830
Residuals	24	365359	15223		

◊: degrees of freedom

¥SS: sum of squares

♦MS: mean squares

**: highly significant

***: very highly significant

Table 3. Linear model (logit transform) ANCOVA comparing the covariates: curved carapace length (CCL) and season to the response variables: A. Percentage volume (V_d *c*.macroalgae); B. Percentage volume (V_d *c*.animal prey).

	df ◊	SS ¥	MS ♦	*F*	*P*-value
A. V_d *c*.macroalgae					
CCL	1	3.742	3.742	0.662	0.424
Season	3	13.590	4.530	0.802	0.505
Residuals	24	1833581	76399		
B. V_d *c*.animal prey					
CCL	1	0.227	0.227	0.034	0.855
Season	3	9.785	3.262	0.489	0.693
Residuals	24	160.003	6.667		

◊: degrees of freedom

¥SS: sum of squares

♦MS: mean squares

4.5 Discussion

Understanding the habitat use of endangered marine species across their range is an important facet of conservation biology because it supports the development of effective conservation measures that encompass the extent of a population's range (Block et al. 2011). In this context, diet component data collected from 34 stranded or incidentally captured green turtles provides evidence that supports the hypothesis posited by Godoy et al (2016) that green turtles recruit and take up residence into the neritic habitats of New Zealand's northern region. In total, 85% green turtles examined in this study had benthic digesta in the anterior portion of the gut (crop and stomach) indicating they had been foraging recently in inshore neritic habitats. In addition, pelagic prey species were absent from the anterior tract, providing further evidence that those examined had recruited into nearshore waters and had transitioned to a benthic feeding strategy. The data also revealed that green turtles foraging in New Zealand continue to do so throughout the year, even during winter when water temperatures are lowest. Although a slight reduction in the average total diet volume consumed during winter was observed, there was no significant seasonal effect on the total or percentage volume of diet intake. Six necropsied turtles initially found stranded (alive or dead) or incidentally captured during winter, had fresh digesta in their crop and stomach indicating these animals had been recently feeding. This suggests that green turtles in New Zealand continue to forage throughout the year despite SST decreasing to approximately 14°C during winter (Godoy et al. 2016; Chapter 1). The data also show no significant difference in the proportion of major diet components (macroalgae and animal prey) consumed seasonally, indicating that green turtles are utilizing similar food resources throughout the year. Although winter temperatures are known to cause compensatory behavioural changes in marine turtles (Hochscheid et al. 2005), foraging continues by some individuals during these periods in the New Zealand aggregation.

Winter feeding in warm temperate habitats of the southwestern Pacific has also been observed in Moreton Bay (ca. 27 °S, 153 °E), Queensland (a permanent warm temperate foraging ground) and at Julian Rocks (ca. 28 °S, 153 °E), New South Wales in temperatures as low as 15°C and 12°C, respectively (Read 1991, Limpus et al. 1994, Read et al. 1996; A. Muyt, pers.obs). However, no study to date has identified a resident foraging aggregation farther south than ~28 °S, thus at ~34-38 °S, the data presented here extends the southern foraging limit for green turtles in the Pacific region.

Diet component analysis revealed green turtles in New Zealand are primarily herbivorous yet frequently consume animal prey. In total, 40 taxa were consumed with 23 taxa identified as major diet components. Macroalgae comprised the largest proportion in terms of frequency and volume ingested, while benthic gastropods featured as a dominant food item. Specifically, the green algae *Codium fragile*, the red algae *Gigatina atropurpurea*, and the gastropod *Pleurobranchaea maculata* were dominant species in the diet overall. Although present in the diet, the seagrass *Z. muelleri* subsp. *novozelandica* and the fruits and leaves of the mangrove *Avicennia marina* were not considered of similar importance to macroalgae or animal prey. The only introduced (non-native) diet item identified was of the bivalve *Musculista senhousia*. This mussel was consumed in >5% in only one individual and present at <5% in only one other individual. This suggests introduced species do not contribute significantly to the diet in New Zealand waters. The relative importance of introduced species in the diet contrasts to Russell and Balazs (2015) where non-native species comprise the bulk of the turtle diet in Kaneohe Bay, Hawaii. However, explicit comparisons cannot be made between the Kaneohe Bay study and this study without first quantifying the abundance and availability of introduced species relative to native species in New Zealand.

Although a negative correlation was found between the consumption of macroalgae and animal matter, which may correlate with an individual's specialisation (Vander Zanden et al. 2013) or represent single feeding bouts, there was no relationship between any of the four major diet categories and CCL. This suggests overall, that green turtles in New Zealand are not transitioning to a strictly herbivorous feeding strategy once they recruit to neritic habitats. Here, they instead continue to feed on a mixed omnivorous diet while resident in neritic habitats. The mixed diets observed for turtles in this study are similar to other areas where omnivory may be a facultative response to food availability, predation risk, differing nutritional requirements between age classes (e.g. juveniles versus adults), or the influence of temperature on digestion physiology (Bjorndal 1980, Bjorndal et al. 1991, Forbes 1996, Davenport 1997, Godley et al. 1998, Read and Limpus 2002, Seminoff et al. 2002b, Amorocho and Reina 2007, 2008, Cardona et al. 2010, Carrión-Cortez et al. 2010, Lemons et al. 2011).

In the New Zealand context, the substantial contribution of animal prey in the diet may be influenced by temperature constraints of a temperate environment upon a poikilothermic reptile. Neritic immature green turtles in New Zealand may be actively supplementing their diet with animal matter to maximise growth during this life stage, and, compensate for a decrease in efficiency of hind gut fermentation in environmental temperatures experienced in New Zealand. Animal matter is more easily digested and contains higher protein content than macroalgae (Bjorndal 1980, Forbes 1996, Vander Zanden et al. 2013, Santos et al. 2015), therefore energy is more easily assimilated and gained during digestion at lower temperatures than can be obtained via hind gut fermentation of vegetal matter. For example, Santos et al (2015) reported that green turtles consumed higher amounts of animal prey below 20 °C to compensate for lower digestive efficiency at this temperature. Similarly, Vander Zanden et al (2013) suggests that neritic juveniles continued to selectively ingest animal prey to maximise growth. In addition, despite clear evidence of year round foraging in this temperate habitat, five new and recent recruits with empty gastrointestinal tracts were found stranded alive between autumn and spring in poor body condition. This suggests that environmental temperature related stress may adversely affect the health status of new recruits transitioning into this temperate developmental habitat during periods of colder SST. Nevertheless, given that histopathological health assessments were not explicitly conducted as part of this study, this observation should be interpreted with caution, particularly as turtles with empty gastrointestinal have been observed in other studies in subtropical habitats (Awabdi et al. 2013, Reisser et al. 2013).

Some aspects of their foraging ecology were also identified from observations during this study. Several turtles contained alternating boluses of monospecific digesta along the anterior tract indicating these turtles were selectively feeding on preferred items representing discrete foraging periods. For example, although several species were not frequently consumed by all turtles across the sample, some individuals consumed large proportions of single species (i.e. with a FOd >50% Vd) indicating singular feeding bouts. Selected species include the red algae *Sarcodia montagneana*, *Pterocladia lucida*, *Gelidium caulacantheum*, and *Psilophycus alveatus*; the green algae *C. fragile*; the seagrass *Z. muelleri* subsp. *novozelandica*; egg masses from the neogastropod *Cominella adspersa*, the gastropod *Bursatella leachii* and *P. maculata*. Notably, in one individual, ca. 70 *P. maculata* (comprising 81% of total digesta) were found in the stomach intermixed with relatively small amounts of *C. fragile* (17%), seagrass (1%) and mangrove fruit (1%). Selective feeding bouts of alternating or differential feeding has been observed in other diet studies (e.g. Balazs 1980, Read 1991, Forbes 1996, Brand-Gardner et al. 1999, Burkholder et al. 2011)(e.g. Balazs 1980, Read 1991, Forbes 1996, Brand-Gardner et al. 1999, Burkholder et al. 2011) and inferred as individuals optimising food availability, abundance, nutritional requirements, or tidal movement.

Although this study provides some key empirical evidence that green turtles do recruit and forage in northern New Zealand’s neritic habitats, drawing inferences about dietary selection based on stranded animals should nevertheless be considered with caution. Diet component analysis of stranded animals only provides a snapshot of the diet and may not accurately reflect the foraging ecology of a healthy free-ranging population. Therefore, further study of free-ranging turtles, their nutrition and food availability in New Zealand would assist with testing the conclusions herein. For example, to address some key aspects of the mixed diet observed in this study, further research into food availability, and the macronutrient components and energy gain of consumed prey may determine if green turtles in this temperate habitat are adequately satisfying their nutritional requirements.

In conclusion, this is the first account of green turtle foraging in New Zealand while extending the geographical southern foraging limit of this species in the Pacific region. Thus, this work provides new baseline understanding of the foraging ecology of this species at the edge of their range in the southwestern Pacific. Data presented here support the hypothesis that green turtles recruit and take up residence into the neritic habitats of northern New Zealand. Specifically, green turtles feed primarily on Chlorophyte and Rhodophyte macroalgae yet supplement their diet with a substantial proportion of benthic gastropods. Results highlight the potential importance of algal-dominated reef habitats and seagrass meadows (sheltered harbours and estuaries) for the endangered green turtle in New Zealand. More broadly, this study provides insight into the diet of a large poikilothermic marine reptile and the potential adaptive foraging strategies employed to compensate for environmental temperature constraints experienced in a temperate habitat.

Chapter 5 **Anthropogenic impacts on green turtles (*Chelonia mydas*) in New Zealand**

5.1 Abstract

Conservation strategies to sustain endangered green turtle (*Chelonia mydas*) populations must categorise and mitigate a range of anthropogenic impacts affecting foraging aggregations. This study provides first insights into some of the adverse effects of anthropogenic activities on endangered green turtles within New Zealand. Gross necropsies were conducted on 35 immature and sub-adult green turtles found stranded in northern New Zealand between 2007 and 2013. Results show 54% (*n* = 19) of examined individuals exhibited anthropogenic related trauma, including 63% (*n* = 12) of these having ingested synthetic marine debris. Soft plastics (i.e. single-use food packaging, plastic bags), and white and clear/translucent items were the most predominant items ingested. No correlation between curved carapace length (CCL) and the volume or number of synthetic debris items ingested was observed. Propeller strike injuries were identified in 26% (*n* = 5) of necropsied turtles exhibiting human related effects, while 10% (*n* = 2) exhibited evidence of incidental capture in recreational fishing activities. Importantly, within New Zealand waters, anthropogenic effects predominantly associated with plastic ingestion are impacting the green turtle aggregation, and may be a major cause for strandings of immature and sub-adult green turtles in this region. Consequently, the threats identified in this study should be considered when developing population specific conservation strategies, as green turtles at this foraging ground originate from several genetically distinct stocks across the Pacific region.

5.2 Introduction

In recent history, human activities have led to a substantial decline in marine biodiversity worldwide (Lewison et al. 2004, Crowder and Norse 2005, Pereira et al. 2012). Today, overexploitation and harvesting, bycatch, habitat loss and degradation, pollution, and climate change continue to threaten marine biota globally (Derraik 2002, Newson et al. 2009, Block et al. 2011, Burrows et al. 2011, Gilman 2011). In addition, of increasing concern is the significant and wide-ranging environmental impact of synthetic marine debris, and in particular the persistent adverse effect of plastic pollution (Gregory 2009, Law et al. 2010, Carson 2013). Species particularly affected by anthropogenic impacts include highly migratory and long-lived taxa such as sharks, marine mammals, seabirds and marine turtles because they exhibit life history traits that render them susceptible to anthropogenic decline and delayed population recovery (Lewison et al. 2004, 2014, Gilman et al. 2008, Anderson et al. 2011, Block et al. 2011, Clarke et al. 2014, Huang 2014).

Marine turtles are a vertebrate guild distributed throughout tropical and temperate regions worldwide, generally between 40° N and 40° S (Pritchard 1997). These highly migratory species exhibit complex life history patterns that encompass coastal nesting areas, neritic foraging grounds, oceanic habitats, and cryptic migratory pathways (Balazs 1976, Hirth 1997, Lohmann and Lohmann 1998, Bolten 2003, Luschi et al. 2003, Boyle and Limpus 2008). Accordingly, marine turtles are exposed to numerous anthropogenic effects across their distributional range and life cycle (Eckert 1995). Thus, although historically abundant, six of seven species have experienced significant declines globally and are now considered ‘Threatened’ under the Red List of the International Union for Conservation of Nature (Seminoff 2004, Abreu-Grobois and Plotkin 2008, Mortimer and Donnelly 2008, Wallace et al. 2013, Casale and Tucker 2015). Consequently, knowledge of the impact of anthropogenic threats on these threatened marine species across temporal and spatial scales is a critical component in the conservation management of marine turtles worldwide (Wallace et al. 2011, Koch et al. 2013).

Marine turtles occupying coastal foraging habitats are at risk of adverse anthropogenic effects including fisheries bycatch, entanglement in and ingestion of marine debris, and vessel collision (Denkinger et al. 2013, Schuyler et al. 2013, Wallace et al. 2013, Lewison et al. 2014, Nelms et al. 2016). Marine turtles ingest synthetic marine debris inadvertently if mixed or attached to natural diet items, or if mistaken for natural prey or forage (Carr 1987, Hoarau et al. 2014, Casale et al. 2016). In a recent study, Schuyler et al. (2012) reported neritic foraging marine turtles selectively consumed soft clear and white plastics, which has been posited to resemble their natural prey such as jellyfish (Carr 1987, Bugoni et al. 2001). In this regard, Plotkin and Amos (1990) suggested that small turtles (particularly pelagic stage juveniles) were more likely to ingest plastics, while older neritic phase sub-adults and adults exhibited a size (age) correlated decrease in plastic consumption. Conversely, Tomás et al. (2002) concluded the volume of plastic ingested correlated with an increase in curved carapace length (CCL) in loggerhead turtles (*Caretta caretta*) in the Mediterranean. Ultimately, for all turtles, debris can accumulate and obstruct, harm or cause inflammation of the digestive tract, leading to reduced digestive ability, diet dilution, reduced fitness and even possible mortality (Bjorndal et al. 1994, Casale et al. 2016, Nelms et al. 2016, Schuyler et al. 2016).

Entanglement in marine debris (including discarded or lost fishing gear) and bycatch in fisheries activities poses a significant threat to marine turtles worldwide (Laist 1997, Lewison et al. 2004, Wallace et al. 2013, Clarke et al. 2014). For example, entanglement in lost or discarded ‘ghost nets’ is a major source of mortality for several species of marine turtle in Northern Australia (Jensen et al. 2013, Wilcox et al. 2013), while incidental capture of marine turtles in pelagic longline, trawl, and coastal gillnet fisheries have also been widely reported (Crowder et al. 1995, McCracken 2000, Robins et al. 2002, Tomás et al. 2008, Donoso and Dutton 2010, Wallace et al. 2013). In addition, for many air breathing vertebrates (e.g. marine mammals and turtles), vessel related injuries such as propeller strike and blunt force trauma caused by vessel collision may also represent a major cause of mortality (Work et al. 2010). Consequently, in developed coastal regions where vessel traffic is high, marine turtles are at increased risk of vessel collision which often cause debilitating if not fatal injuries (Work et al. 2010). For instance, vessel strike injuries accounted for 9% of all stranded marine turtles on the Mediterranean coast of Spain (Tomás et al. 2008), while in southeast Florida, up to 60% of stranded loggerhead turtles exhibited injuries consistent with propeller strike (Work et al. 2010). In Queensland, Australia, between 1999 and 2002, 12–16% of annual stranding records of green (*Chelonia mydas*) and loggerhead turtles were due to vessel related injuries (Hazel and Gyuris 2006).

Given the complex life history, wide ranging distribution, and significant anthropogenic threats marine turtles are exposed to throughout their lives, conservation managers need an understanding of the types and extent of anthropogenic mortality to formulate appropriate population specific mitigation (Seminoff et al. 2002a, Wallace et al. 2011, Wallace et al. 2013). For instance, although a single human activity may not have a substantial negative impact on a marine turtle population overall, the cumulative negative effect of several different human activities in combination (e.g. vessel collision, plastic pollution and bycatch), may cause a significant decline in a population (e.g. Limpus 2008a, 2008b). The risk of cumulative negative effects increases further near densely human populated regions that overlap marine turtle aggregations, such as coastal foraging grounds. Here, marine turtles are often exposed to a suite of anthropogenic effects which cumulatively increases the risk of population decline overall (e.g. Gardner et al. 2003, Hazel and Gyuris 2006, Barnes et al. 2009, Work and Balazs 2010). However, monitoring free-ranging marine turtles at coastal foraging grounds is logistically challenging and therefore often overlooked (Seminoff et al. 2003, Chaloupka et al. 2008, Nelms et al. 2016). Further, assessing anthropogenic impact *in situ* is more problematical given that it is a rare event, hence often goes unobserved (McCracken 2004). Therefore, examinations of stranded turtles from coastal foraging grounds can be used to elucidate key threats to a foraging aggregation which could otherwise go undetected (Chaloupka et al. 2008, Cole et al. 2011, Koch et al. 2013).

In New Zealand, recent research has identified a temperate neritic foraging aggregation of immature green turtles (Godoy et al. 2016; Chapter 2 and 4). The aggregation comprises a mixed stock foraging ground with links to several genetically distinct management units that span the Pacific Ocean region (Chapter 3). Although bycatch data suggests green turtles are at risk of incidental capture in commercial fisheries waters around New Zealand (Godoy 2016), other potential threats have not been investigated. Therefore, by undertaking post mortem examinations of stranded animals, the aim was to investigate anthropogenic effects on green turtles in New Zealand. Specifically, stranded turtles were assessed to: 1. assess the ingestion of synthetic marine debris and ascertain whether a correlation between size (CCL) and volume exists; 2. determine the type and colour of debris predominantly ingested; 3. identify evidence of entanglement, vessel collision and bycatch; 4. describe any other significant contributing factors to green turtle mortality.

5.3 Methods

Between 2007 and 2013, a total of 48 stranded green turtles were reported in New Zealand (see Chapter 2: Figure 1). Twenty six turtles were found dead while 22 were initially observed alive (of which 21 subsequently died and one was rehabilitated and released). In total, 35 were recovered and assessed for anthropogenic impacts using standard gross necropsy techniques (Wolke and George 1981, Flint et al. 2009). Because the focus of post-mortem examinations was to ascertain life history, carcasses examined herein were assessed only at the gross level. Stranding date, location, and standard curved carapace length (CCL, ±0.1 cm) measurements were recorded to assess the size (age) class of the sample (Limpus et al. 1994, Bolten 1999). Sex and maturity status was determined by *in situ* examination of the gonads and associated ducts (following the criteria of Rainey (1981), and Limpus and Reed (1985)). Where possible, sex and maturity status was confirmed histologically, with turtles categorised as immature or mature (Limpus et al. 2005, Meylan et al. 2011). Gross lesions, abnormalities and other potentially relevant indicators were recorded, measured and photographed.

To investigate the extent of synthetic debris ingestion, and where in the gastrointestinal tract debris had accumulated, the complete tract was removed and divided into the anterior (oesophagus and stomach) and posterior (small and large intestine) sections. The gut was examined for areas of impaction, haemorrhaging, or lesions caused by ingested synthetic debris (as per Flint et al. 2009). The location of any impaction or related observation within the gastrointestinal tract was recorded and photographed. The entire contents were then collected and rinsed through a 0.5 mm fine mesh sieve. Diet items (food digesta) and natural debris (e.g. wood, pumice, feathers) were separated for diet component analysis (Chapter 4), while synthetic debris was washed and dried at room temperature for processing (Hanke et al. 2013).

For each turtle sampled, debris items were identified and categorised according to type as described in Schuyler et al. (2012): hard plastic, soft plastic, synthetic rope/twine, non-synthetic rope, fishing items, balloons, other rubber, foam, other (e.g. tar/oil, metal, glass, cloth); and according to colour: white, clear/transparent, red, orange, yellow, green, blue, brown, black, other. The total number of each type and colour of debris within each turtle was recorded, weighed (± 0.01 g), and volume measured using the volume displacement method with alcohol in a graduated cylinder (± 0.1 ml) (Schuyler et al. 2012, Santos et al. 2015). The total frequency of occurrence (%*FO*) of each type and colour was subsequently quantified as:

$$\%FO = \frac{\text{(Number of turtles in which debris item was observed)}}{\text{Total number of turtles with ingested debris}} \times 100$$

The relative percentage (%*R*) of each type and colour within each turtle was quantified as:

$$\%R = \frac{\text{(Number of items of each type.colour ingested)}}{\text{Total number of items ingested by each turtle}} \times 100$$

To examine whether smaller turtles consumed greater relative amounts of synthetic debris, a *t*-test was applied to test the null hypothesis that there was no significant difference in CCL between turtles which ingested synthetic debris and those that had not. Linear regression analysis was used to investigate the relationship between CCL and the number and volume of synthetic debris items ingested per turtle. The variables CCL, number of pieces, and volume of synthetic debris were log transformed for regression analysis and alpha was set at 0.05.

During gross necropsy, turtles were examined for evidence of entanglement in marine debris, fishing interaction or vessel strike injuries. Categories were defined as 1. entanglement: turtles presented with evidence of interaction in either discarded fishing gear or other type of marine debris (i.e. linear rope marks, external lesions and indentations); 2. fishing interaction: turtles presented with evidence of interaction active fishing related gear (e.g. set nets, crayfish pots) or hooks were found embedded externally (e.g. mouth cavity or flipper) or internally (e.g. swallowed hook and line); 3. vessel/ propeller strike: identified as catastrophic blunt trauma (e.g. fractures, haemorrhaging), as multiple evenly spaced parallel lacerations (propeller), or single linear laceration (skeg) (Norem 2005, Flint et al. 2009, Work et al. 2010, Martinez and Stockin 2013). For this study, a catastrophic injury was defined as any wound that fractured or penetrated the carapace/body, compromising the coelomic cavity, thus causing immediate or delayed mortality via infection (Work and Balazs 2010).

For each turtle examined, the likely cause or significant contributing factor to mortality was determined based on the most significant and severe finding (Chaloupka et al. 2008). For example, where a catastrophic vessel collision injury was identified and no other external or internal gross pathology observed, vessel strike was considered the most likely cause of mortality. Given the small sample size overall, seasonal, sex, and size class effects could not be statistically tested, therefore only a descriptive summary for each factor is presented.

5.4 Results

All turtles were found stranded (alive or dead) on the coastline of the North Island between c. 38°–34° S (see Chapter 2: Figure 1). Stranded turtles were observed year round with seasonal distribution consistent throughout the year, yet with marginally higher stranding numbers during austral spring (n = 15) compared to summer (n = 6), autumn (n = 10) and winter (n = 4) (Godoy et al. (2016) and Chapter 2). The size distribution (CCL) of all assessed turtles encompassed a range of 37.3–94.6 cm (μ = 51.9 cm, SD = 12.3, n = 35). Where sex could be identified (n = 31), all were immature juveniles to large sub-adults of both sexes (19 female, 12 male). Of the 35 turtles assessed for anthropogenic trauma, 54% (n = 19) exhibited evidence of human impacts. Twelve turtles (34%) had ingested synthetic debris (μ = 49.6 cm CCL, SD = 11.3, range 37.3-66.4 cm), while 23 individuals (66%) were devoid of synthetic debris in the gastrointestinal tract (μ = 53.1 cm CCL, SD = 13.6). Between these two groups, there was no significant difference in mean CCL (t = 0.86, df = 28, P = 0.40).

All turtles (n = 34) were found to contain natural digesta in their gastrointestinal tract, except for one individual (ID: 064) which was also devoid of synthetic debris. In total, 791 pieces of synthetic debris were ingested by 12 turtles, with a mean of 65.9 pieces per turtle, although ingestion rate was highly variable between individuals (SD = 128.2, range = 1-432). Similarly, ingested volumes were also variable between individuals (0.1-45 ml, μ = 8.3 ml, SD = 15.9). Linear regression analysis revealed no significant correlation between CCL and the number of pieces (r^2 = 0.21, t = 1.64, df = 10, P = 0.13) or volume (r^2 = 0.08, t = 0.93, df = 10, P = 0.37) of synthetic debris ingested.

Frequency of occurrence and relative percent abundance revealed soft plastic was the type most frequently consumed (%FO = 91.7) and in the largest relative amounts (%R = 51.0; Table 1; Figure 1A, 1B). In addition, white (%FO = 66.7, %R = 16.6) and clear/translucent categories (%FO = 83.3, %R = 54.1) were most frequently consumed and in the highest relative quantities (Table 2; Figure 1A, 1B). Debris types identified included single use plastics such as food packaging, balloons and bags, while fishing line and synthetic ‘soft-bait’ lures were also recorded (Figure 1A, 1B; Figure 2). Of the 12 turtles with ingested synthetic debris, nine contained debris only in the posterior tract (small and large intestines), while three contained debris in the anterior (stomach) and posterior tract. Four turtles contained significant amounts of ingested synthetic debris leading to severe impaction of the gastrointestinal tract, and perforation of the intestinal wall in one individual. In addition, one turtle had ingested fishing line measuring 122.5 cm which became lodged in the small intestine, causing severe plication and granulomatous inflammation of the surrounding tissue. No turtles were found to contain synthetic debris only in the anterior portion of the gastrointestinal tract.

Of the 35 turtles assessed for vessel-related injuries, five (14%) exhibited clear evidence of catastrophic propeller strike injuries (e.g. Figure 3). A further two turtles (6%) had been categorised as incidentally caught given that recreational hooks were embedded in the oesophagus anteriorly between the tongue and the glottis (Figure 4). Of the seven individuals showing evidence of propeller strike and recreational bycatch, none contained ingested synthetic debris. No turtles exhibited injuries or marks consistent with entanglement either in active or discarded fishing gear, or other debris. In the absence of forensic examination, the cause of death could not be conclusively determined, however, significant contributing factors were evident in 11 cases. Four turtles exhibited significant gut impaction or intestinal plication; five were presented with catastrophic propeller strike injuries; and two inferred as been incidentally captured in recreational fishing activities.

Table 1. Frequency of occurrence (%*FO*) and relative percentage abundance (%*R*) of synthetic marine debris types observed in the gastrointestinal tract of stranded immature and sub-adult green turtles in New Zealand (*n* = 12).

Debris type	n	%FO	%R
Soft plastic	11	91.7	51.0
Plastic	8	66.7	28.9
Hard plastic	5	41.7	18.1
Balloons	2	16.7	1.4
Fishing items	1	8.3	0.1
Other rubber	3	25.0	0.5
Total			100

Table 2. Frequency of occurrence (%*FO*) and relative percentage abundance (%*R*) of synthetic marine debris colours recorded in the gastrointestinal tract of stranded immature and sub-adult green turtles in New Zealand (*n* = 12).

Debris colour	n	%FO	%R
Clear/translucent	10	83.3	54.1
White	8	66.7	16.6
Blue	6	50.0	9.1
Black	5	41.7	5.7
Green	4	33.3	6.8
Yellow	4	33.3	3.8
Orange	2	16.7	0.8
Red	1	8.3	0.1
Brown	3	25.0	1.8
Other	3	25.0	1.4
Total			100

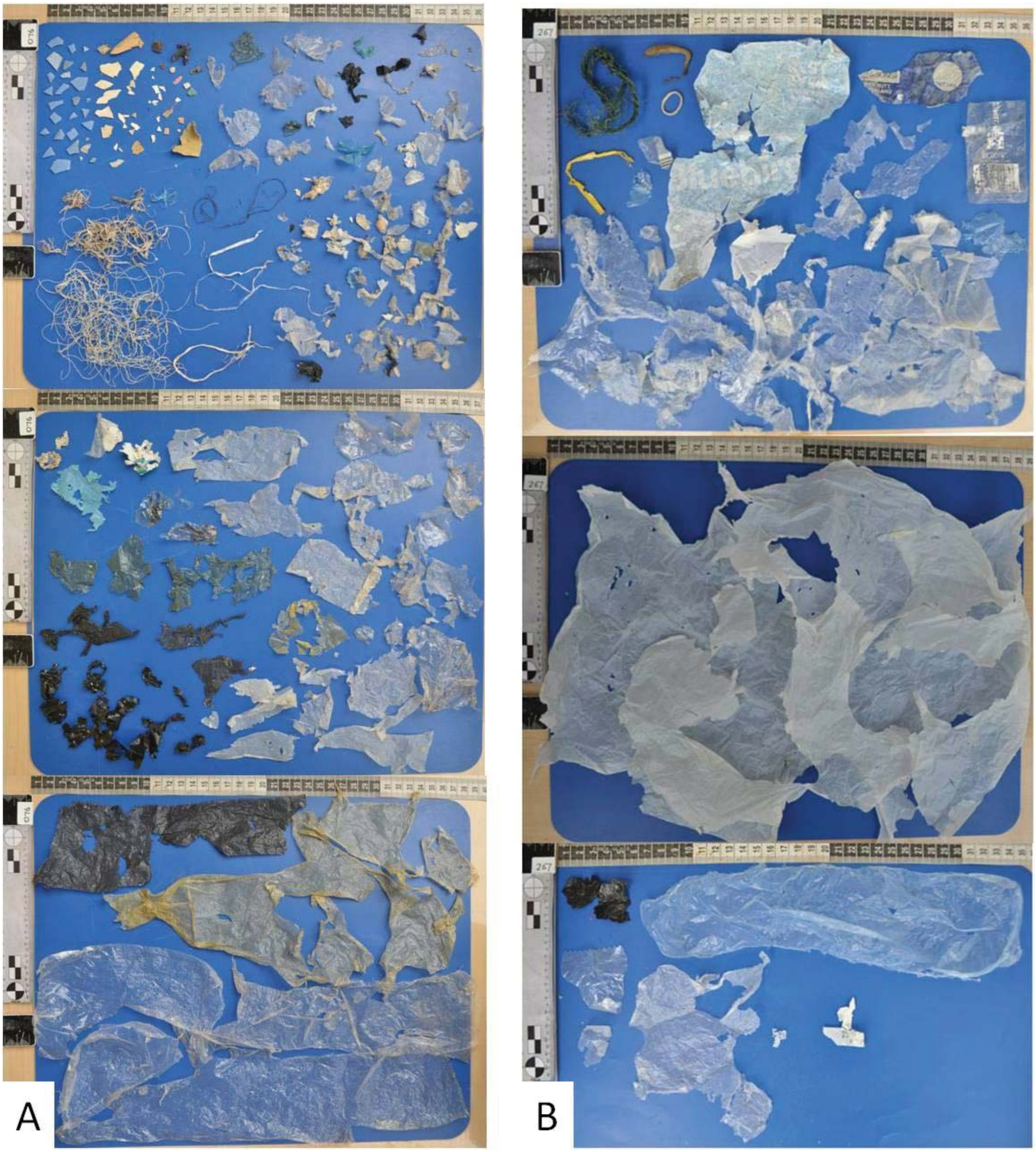

Figure 1. Examples of ingested synthetic debris by two stranded turtles in New Zealand. Turtle A (ID: 076, 43.7 cm CCL, 7.7 kg) and turtle B (ID: 267, 66.4 cm CCL, 28.3 kg) exemplifying the prevalence of soft, white and clear/translucent plastics ingested.

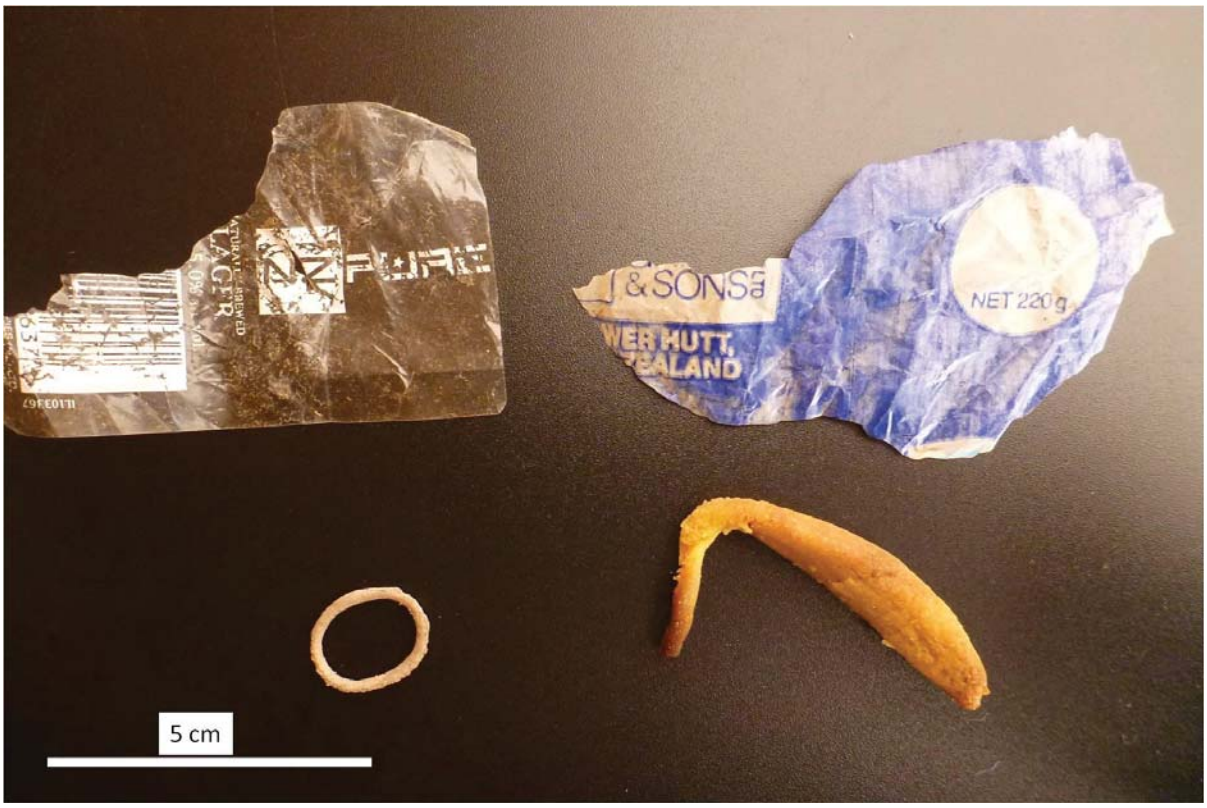

Figure 2. Single use plastic food packaging of New Zealand branded products, a balloon and ‘soft-bait’ lure found in the digestive tract of a stranded green turtle in the Waitemata Harbour, Auckland.

Figure 3. Examples of two stranded green turtles exhibiting catastrophic propeller strike injuries. Note the evenly spaced parallel lacerations causing severe fracture and penetration of the carapace in the turtle A (ID: 094, 77.3 cm CCL) and turtle B (ID: 267, 76.2 cm CCL).

Figure 4. Recreational fishing hooks retrieved from two immature green turtles found stranded in Waitemata Harbour, Auckland; turtle A (ID: 094, 63.2 cm CCL) and turtle B (ID: 365, 45.8 cm CCL). Both hooks were embedded in the oesophagus anteriorly between the tongue and the glottis.

5.5 Discussion

Post-mortem examinations of stranded animals can provide insight into sources of anthropogenic impacts which may otherwise go undetected in a population (Chaloupka et al. 2008, Stockin et al. 2008, Flint et al. 2009). For instance, Casale et al. (2016) suggest that stranded turtles are the best representative of neritic coastal foragers, thus can reveal important information on the threats and risks to resident populations (Chaloupka et al. 2008). In turn, information derived from such studies can highlight specific population level impacts and inform future mitigation and conservation strategies (Crowder et al. 1995, Wallace et al. 2011, Casale et al. 2016, Nelms et al. 2016). Accordingly, in this study, gross necropsies were conducted on stranded green turtles found on New Zealand’s northern coastline between

2007 and 2013 to identify and describe the anthropogenic impacts that may threaten the globally endangered green turtle in New Zealand waters. We consider the stranded turtles examined in this study were representative of the neritic foraging aggregation found in New Zealand because the demographic structure sampled reflects the broader population structure reported for this aggregation by Godoy et al. (2016; Chapter 2).

Overall, 54% (*n* = 19) of stranded turtles examined herein were symptomatic of anthropogenic impacts, suggesting human activities may have a significant adverse effect on the green turtle aggregation in New Zealand. Observed impacts include the ingestion of synthetic marine debris (of terrestrial and marine origin), vessel strike injuries, and incidental capture in recreational fishing activities. Although results identify the types of anthropogenic impacts that green turtles are exposed to in New Zealand, they do not quantify the absolute number of human induced turtle mortality in this aggregation. This is because stranding probability is influenced by extrinsic factors such as oceanographic conditions, distance from shore and mortality type (Chaloupka et al. 2008, Koch et al. 2013). Hence, the likelihood of stranding ashore (and thus available for collection and examination) has been estimated as low as 10-20%, even in nearshore waters (Koch et al. 2013). Considering this, it is expected the actual number of human induced turtle mortality observed in this study to be substantially higher, and therefore should be treated as a minimum estimate when evaluating population wide impacts.

5.5.1 Synthetic debris ingestion

The levels of synthetic debris ingestion identified here were similar to the amounts reported for benthic foraging green turtles in Australia (Schuyler et al. 2012) and fell mid-range within the levels reviewed by Nelms et al. (2016) from studies worldwide. Similarly to studies at other foraging grounds in the Mediterranean (Casale et al. 2016), southern Brazil (Bugoni et al. 2001) and eastern Australia (Schuyler et al. 2012) for example, soft plastics and white or clear/translucent items were also the most prevalent debris types consumed. It is unclear whether the items consumed by green turtles in New Zealand proportionally reflect the quantity of marine debris discharged (and therefore available for incidental consumption) or whether they are selectivity consumed i.e. mistaken for natural forage or prey. However, Schuyler et al. (2012) reported that when compared to marine litter abundance (as a measure of availability) in eastern Australia, neritic turtles selectively consumed white and clear soft plastics over hard and coloured items. In addition, of particular note, was a prevalence of single use plastics (e.g. food packaging and plastic bags) found in the gastrointestinal tracts of several turtles. The pervasiveness of single use plastics observed in this study and others (e.g. Santos et al. 2015) is concerning given that this category of plastic has been shown to be the fastest growing component of waste today (Moore 2008).

The adverse impact of discarded land-based plastic waste on vulnerable marine species is often further intensified where large urban centres are located near coastal areas (Moore 2008). This is because levels of generated waste are substantially higher in densely populated urban centres in comparison to rural and undeveloped regions, thus significantly higher volumes of plastic debris may be discharged, either purposely or inadvertently, out to sea (Derraik 2002). For marine turtles occupying neritic foraging grounds in close proximity to these highly urbanised areas, the impact of discharged waste may therefore have a considerable negative effect (Nelms et al. 2016). In addition, the physical characteristics of the region encompassing a coastal foraging ground e.g. in enclosed seas where the flushing effect of currents and tidal flow is limited, may lead to higher concentrations of plastic debris accumulating (Barnes et al. 2009). In this context, given that Auckland is New Zealand’s largest urban and industrialized centre (Bayley and Goodyear 2004), and is located adjacent to the semi-enclosed coastal sea of the Hauraki Gulf, relatively high concentrations of marine litter have been recorded (Gregory 1991, Backhurst and Cole 2000, Young and Adams 2010). This northeastern region of the North Island also overlaps with a core neritic habitat for green turtles in New Zealand (Godoy et al. 2016; Chapter 2). Therefore, it is likely a higher proportion of green turtles occupying New Zealand neritic habitats would be at risk of ingesting synthetic debris. In addition, given the lack of correlation between the size (age class) of turtles examined and the volume or number of pieces ingested, data suggest that the risk of debris ingestion is equal across the aggregation; which is in accordance with other studies of similar sized neritic foraging green turtles (e.g. Bugoni et al. 2001, Schuyler et al. 2012). Overall, the size range of turtles that ingested synthetic debris was also not significantly different to those that had not, thus providing no evidence of intraspecific bias.

5.5.2 Vessel collision and fisheries interactions

Since neritic stage green turtles often take up residence to occupy shallow embayments, estuaries and harbours (Hirth 1997, Limpus et al. 2005, Koch et al. 2007, Bresette et al. 2010), they are also at risk of vessel collision injuries and bycatch, particularly in areas adjacent to densely populated coastal regions (Limpus et al. 1994). These results support this, where almost all turtles presented with vessel collision injuries (propeller strike) or captured in recreational fisheries, were recovered near Auckland (Waitemata) or Whangarei harbours. These highly urbanised regions have high levels of recreational and commercial vessel traffic and have been shown to also have higher incidences of fatal vessel collisions for marine mammals (Martinez and Stockin 2013, Dwyer et al. 2014). Although there was evidence of incidental capture in recreational fisheries, commercial fisheries interactions were not identified. Despite this, recent research suggests that green turtles occupying this northeastern region of New Zealand are at risk from inshore commercial fisheries activities (Godoy 2016; Appendix 3). In addition, entanglement was not identified as a cause of injury or mortality in this study, however, mortality caused by entanglement (mainly via asphyxia/drowning) in fishing nets is difficult to identify due to an absence of visible lesions and therefore, is often underestimated (Bugoni et al. 2001). Despite a lack of evidence of entanglement of green turtles in this study, the entanglement of other marine species in fishing gear in New Zealand has been observed, including leatherback turtles (*Dermochelys coriacea*) (Godoy et al. 2011), marine mammals (Slooten and Dawson 1995, Boren et al. 2006, Stockin et al. 2009) and seabirds (Abraham and Thompson 2011, Bell 2014). Furthermore, entanglement in active or discarded fishing gear is a significant issue for marine turtle mortality in other regions (e.g. northern Australia and Mediterranean) and therefore its potential risk in New Zealand cannot be overlooked (Nelms et al. 2016, Schuyler et al. 2016).

5.5.3 Causes of mortality

Conclusively diagnosing the cause of mortality in stranded marine animals is difficult and requires comprehensive histopathological post-mortem examinations of fresh carcasses (Chaloupka et al. 2008, Flint et al. 2009, Stockin et al. 2009). It should be noted therefore, comprehensive histopathological or toxicological samples were not collected during gross necropsies, and therefore other effects (e.g. disease, anthropogenic related chemical toxicity) were not examined here. While this was not logistically plausible in this study, in several cases reported herein, gross necropsies still revealed incidences of ingested synthetic debris, incidental capture and catastrophic propeller strike trauma severe enough to conclude these factors as the leading probable cause of mortality. For example, propeller strike was deemed the leading probable cause of mortality in at least two cases given that (1) ante-mortem body condition was good (absence of muscle or adipose atrophy), (2) there was an absence of any obvious gross pathology (abnormalities, lesions, epibiont or parasite load), (3) significant hemorrhaging and trauma was evident around the wound sites indicating the turtles were alive at the time of impact. In addition, fresh digesta in the stomach and crop suggested they had been foraging just prior to death. Therefore, evidence suggests that in both cases, these turtles died as a consequence of the injuries sustained.

In relation to mortality due to ingested marine debris, four turtles exhibited severe gut impaction of the intestinal tract due the accumulation of synthetic debris. This resulted in severe inflammation, perforation or plication of the intestinal tract, leading to the conclusion (based on gross analysis) that these turtles most probably died as a consequence of ingesting marine debris. Such an inference is plausible given that Santos et al. (2015) quantified that amounts as low as 0.5 g are sufficient to block the digestive tract and cause death in juvenile turtles. In their study, debris-induced mortality was estimated at 39.4% compared to 42% reported here. Other lesser known implications of the ingestion of synthetic debris by marine turtles are dietary dilution (McCauley and Bjorndal 1999) and the sub-lethal effects of desorbed or leached organic contaminants from plastics (Moore 2008, Engler 2012, Nelms et al. 2016). This issue is compounded because synthetic debris has been shown to accumulate and remain in the digestive tract for prolonged periods in marine turtles (Plotkin and Amos 1990, Casale et al. 2016) and therefore, increases the likelihood of toxic contamination (Teuten et al. 2009). Thus, investigations into the absorption and sub-lethal impacts of organic contaminants in green turtles in New Zealand would also supplement this area of research internationally. Overall, to conclusively investigate the findings described herein, detailed histopathological post-mortem examinations on fresh carcasses would be beneficial. Furthermore, health assessments of live free-ranging green turtles in coastal habitats would provide population specific baseline data such as body condition indices for comparison with stranded turtles.

5.5.4 Conclusion

This study provides the first description of the predominant sources of anthropogenic impacts affecting green turtle populations within northern New Zealand. The range and magnitude of impact observed herein reflects the threats reported globally, with ingested synthetic debris and propeller strike being the most important precursor to stranding and mortality. However, the present work suggests that the risk to such impacts will be considerably higher for turtles inhabiting neritic habitats adjacent to densely populated urban centres of northeastern New Zealand. Importantly, the focal aggregation comprises a mixed stock foraging ground with links to genetically distinct populations from across the Pacific Ocean (Chapter 3 and 4). Thus, this study identifies several adverse human impacts that may impact those distant source populations of this wide ranging endangered species. In turn, this underscores the need to consider all potential threats across a populations' entire distributional range and congruent jurisdictions to appropriately scale conservation strategies.

Chapter 6 **General discussion**

6.1 Summary of research findings

Ecological knowledge about threatened species is fundamental to defining effective conservation territories and strategies (Campbell and Godfrey 1994). The green turtle (*Chelonia mydas*) is a highly migratory and widely dispersed marine poikilotherm. Adverse human impacts have led to significant declines in abundance of this species worldwide, thus, it is now listed as globally ‘endangered’ in the IUCN Red List of Threatened Species (Seminoff 2004). Hence, research into the ecology of the green turtle across its distributional range is beneficial to their conservation. In New Zealand, because information about the green turtle is almost non-existent, this thesis constitutes a comprehensive and cohesive baseline understanding of the biology and ecology of the green turtle in this region. Thus, this work represents a major contribution to our understanding about green turtles in New Zealand, while substantially increasing our knowledge of this species’ distribution, range, diet, regional connectivity and anthropogenic threats in the South Pacific Ocean. Further, evidence presented herein extends the southern range of the green turtle in the Pacific Ocean, thus challenges the conventional paradigm that the green turtle is constrained by the 20 °C isotherm. Accordingly, this work also provides fundamental baseline data for investigating the green turtles’ response to climate-mediated temperature shifts in light of rising global temperatures.

A key focus of the research was to test the hypothesis that the presence of green turtles in New Zealand was incidental and this temperate habitat was not adequate for sustaining this species year round. To achieve this, several integral lines of empirical enquiry were examined in relation to different aspects of green turtle biology in New Zealand. Sighting, stranding and incidental capture data were analysed to determine if green turtles were found year round despite winter sea surface temperatures (SST) decreasing to what is generally considered below their thermal tolerance (Chapter 2). Systematic gross necropsies of green turtles were undertaken to describe the population structure (Chapter 2), and in turn, results were examined to investigate whether their occurrence was due to stochastic events or reflected a natural dispersal and settlement pattern (Chapter 2). To explore the possibility of residency in New Zealand, and thus provide support for rejecting the hypothesis that green turtles are incidentally washed ashore from oceanic habitats; diet component analyses were undertaken to investigate whether green turtles are foraging in neritic habitats (Chapter 4). Genetic analyses were conducted to determine the stock origin of the green turtle aggregation and investigate regional connectivity, dispersal and migratory pathways (Chapter 3). In addition, gross necropsies were also used to investigate anthropogenic impacts upon the aggregation of this globally endangered species in New Zealand (Chapter 5). Ultimately, this work provides a cohesive baseline that addresses several key ecological aspects of green turtles in New Zealand, thus provides a foundation upon which further research may be conducted.

Previous work into the presence of marine reptiles in New Zealand waters describes the occurrence of green turtles as ‘waifs or strays’ (McCann 1966), occasional visitors (McCann 1966), or ‘stragglers’ likely to originate from Australia or further north (Gill 1997). This prevailing inference is plausible given that the average SST from autumn to spring in New Zealand falls outside the preferred thermal envelope for this species (Màrquez 1990, Chiswell 1994, Davenport 1997). In contrast, this Chapter 2 provides empirical evidence to conclude that northern New Zealand constitutes a transitional developmental habitat for post-settlement immature to sub-adult green turtles.

Despite a seasonal SST mean in Northland of 14 °C during austral winter, live free-ranging turtles were observed throughout the year. Although sighting records on the east coast significantly decreased during winter, stranding records remained relatively stable temporally, suggesting that green turtles were still present yet their detection probability was affected in some way. A seasonal reduction in marine observers reporting sightings was postulated as a reasonable effect, yet other explanatory factors were also suggested (1) a movement into warmer water (offshore) during winter in response to seasonally cooler water (Mendonca 1983, Epperly et al. 1995, Ogren and McVea 1995); and (2) green turtles exhibiting a ‘sit and wait’ response during periods of cold SST (Hochscheid et al. 2005). Recent satellite telemetry data into the post-release survival of rehabilitated immature green turtles in New Zealand waters have exhibited both migrations into oceanic habitats adjacent to New Zealand (and their subsequent return) and, ‘sit and wait’ behaviour during winter (D. Godoy, unpubl. data). Thus, although it is highly likely that the seasonal reduction in SST does affect green turtle behaviour in this temperate habitat, it does not appear to preclude their presence year round (Chapter 2).

Sighting, stranding and incidental capture data revealed that the New Zealand aggregation comprises a discrete assemblage of post-pelagic immature juveniles to large sub-adults (μ = 51.9 cm CCL, range 17.6–94.6 cm), and that recruitment occurs at approximately 40.8 cm CCL (range 37.3–44.8 cm; Chapter 2). The observed size at recruitment is similar to that reported for other neritic aggregations in the Pacific Ocean (Balazs 1985, Limpus et al. 2005, Arthur et al. 2008). Given that nesting does not occur in New Zealand, a key facet of this investigation therefore, was to investigate where resident green turtles originate from and the likely dispersal mechanisms that bring them to New Zealand. Gill (1997) proposed that marine reptiles most likely arrive via the Tasman Front (TF) and East Auckland Current (EAUC) in a similar way in which tropical fish species arrive to north-eastern New Zealand (Francis and Evans 1993, Francis et al. 1999). This inference is logical given that New Zealand’s landmass dissects the western boundary of the anti-cyclonic South Pacific sub-tropical gyre system (SPSG), with which the TF and EAUC are associated (Ridgway and Dunn 2003); and, considering the general dispersal patterns of post-hatchling marine turtles (Carr 1980, Bolten 2003). The influence of the SPSG on the dispersal of post-hatchling marine turtles was also proposed by Boyle (2006) and Boyle et al. (2009). They posit that neonate marine turtles emerging from southwest Pacific rookeries (eastern Australia) disperse into the East Australian Current (EAC) and its associated offshore eddies where they remain for several years before recruiting into nearshore neritic habitats. Hence, the population structure observed in this study (Chapter 2) corresponds with a natural dispersal and settlement pattern recognised for green turtles in the southwestern Pacific.

To investigate the dispersal scenarios proposed in Chapter 2, genetic analyses (Chapter 3) were used to determine the genetic stock origin of green turtles in the New Zealand aggregation. Results did indeed indicate 52% of green turtles carried haplotypes commonly reported in southwest Pacific rookeries i.e. southern Great Barrier Reef, Coral Sea, New Caledonia (Chapter 3), supporting the hypothesis of post-hatchlings dispersing via the SPSG and its associated sea surface currents (EAC, TF, EAUC) to New Zealand (Chapter 2, Chapter 3). Although the majority of the sampled aggregation derived from proximate southwest Pacific genetic stocks as would be expected, what makes the New Zealand aggregation unique is the large proportion of individuals (29%, Chapter 3) carrying haplotypes from very distant east Pacific genetic stocks (Revillagigedo, Michoacan, Costa Rica, Galapagos Islands). In this regard, no other published studies have identified east Pacific haplotypes in any other foraging aggregation in the southwest Pacific despite long term research conducted in those areas (Dethmers et al. 2010, Read et al. 2015, Jensen, Bell, et al. 2016).

The unique composition of the New Zealand foraging aggregation (Chapter 4) is reflected by the estimates of genetic diversity which were significantly different to all other breeding stocks and foraging aggregations assessed in this study (Chapter 3). The prevalence of east Pacific individuals in New Zealand also illustrates for the first time, trans-oceanic dispersal of these stocks into the southwest Pacific Ocean. Again, the SPSG and the associated South Equatorial Current (SEC) is the likely oceanographic feature regulating this long distance westward migration. The substantially longer migration of east Pacific green turtles to New Zealand appears reflected in the significantly larger size class of the east Pacific cohort compared to turtles originating from southwest Pacific stocks (Chapter 3). Overall, the New Zealand mixed stock foraging aggregation links 14 genetically distinct green turtle management units from across the Pacific region. Thus, this study highlights the wide ranging population connectivity across the region, and substantially extends the distributional range of east Pacific stocks not previously described. Consequently, conclusions herein should be considered when formulating conservation management of the genetic stocks represented in the New Zealand aggregation.

Upon recruitment into New Zealand, diet component analyses (Chapter 4) suggests green turtles transition to benthic feeding strategy, providing further evidence supporting the hypothesis that green turtles recruit into Northland's inshore waters and reside for a period of time. In addition, necropsied green turtles recovered during winter also contained fresh digesta in their anterior gastrointestinal tracts, providing further evidence that they are capable of remaining active and continue foraging during these periods (Chapter 4). Equivalently, winter foraging at similar SSTs has been observed in Moreton Bay (ca. 27 °S 153 °E), Queensland and at Julian Rocks (ca. 28 °S 153 °E), New South Wales in temperatures as low as 15°C and 12°C, respectively (Read 1991, Limpus et al. 1994, Read et al. 1996; A. Muyt, pers.obs). Ultimately, this work identifies for the first time that green turtles are foraging in New Zealand northern neritic habitats. This new information extends the southern range and foraging limit of the green turtle in the Pacific region. Although the diet reported in this study (Chapter 4) primarily comprised macroalgae, diet component data illustrates that green turtles supplement their diet with a substantial amount of animal prey.

The degree of omnivory observed in this study does not alter with CCL, suggesting green turtles do not ontogenetically transition to a strictly herbivorous diet once they recruit into neritic habitats in New Zealand. Results herein contrast with conventional studies into green turtle diets that show a rapid and definitive transition from epipelagic omnivores to a herbivorous diet of seagrass and macroalgae (e.g. Bjorndal 1980). Results, however, do agree with more recent studies across the green turtle's geographic range illustrating a degree of plasticity both in terms of diet selection and the ontogenetic transition to herbivory (Seminoff et al. 2002b, Cardona et al. 2010, e.g. Awabdi et al. 2013). Thus, results presented in this study provide further evidence that green turtle diet selection and nutrition is more complex and variable than reported in earlier studies, essentially involving a complex interaction between an individual’s nutritional requirements and environmental constraints (e.g. temperature, food availability). It is posited that green turtles actively supplement their diet with animal prey to potentially maximise their growth during the immature life stage while possibly compensating for a decrease in efficiency of hind gut fermentation caused by environmental temperatures experienced in New Zealand.

Despite this study identifying the year round presence of foraging green turtles, supporting the hypothesis that green turtles recruit into neritic habitats of the north Island (Chapter 2, Chapter 4), the absence of mature adults indicates that turtles are departing from New Zealand coastal areas prior to the onset of puberty. It is inferred therefore, that immature green turtles are most likely transitioning through New Zealand as part of broader developmental migration. Developmental migrations (Carr and Caldwell 1956, Carr et al. 1978) and immature-dominated cohorts at coastal foraging grounds (Musick and Limpus 1997, Moncada et al. 2006, Meylan et al. 2011) have been observed in other regions. These developmental migrations have been posited as response to a change in ontogenetic or ecological requirements prior to ultimately settling at an adult foraging ground. In New Zealand, the factors that influence their departure and the timing of departure remain unclear at this stage, and therefore warrant further investigation.

An important facet of the conservation of endangered species is gaining an understanding of the anthropogenic threats they are exposed to across all life stages and habitats occupied (Koch et al. 2013). Developing appropriate management and mitigation measures becomes particularly complex for widely dispersed and long-lived migratory species such as marine turtles (Wallace et al. 2011). This research describes for the first time the type and extent of adverse human impacts and provides evidence that these impacts may be a major cause of stranding for the globally endangered green turtle in New Zealand waters (Chapter 5). Results show that more than half of all examined turtles exhibited signs of human related trauma, including the ingestion of synthetic marine debris, propeller strike injuries and incidental capture in recreational fisheries activities. The ingestion of discarded plastic waste, and in particular soft plastics (i.e., single-use food packaging, plastic bags), and white and clear/translucent items were the most predominant anthropogenic effect observed overall. There was no correlation between size (CCL) of the turtle and the volume or number of synthetic debris items ingested, suggesting that all turtles (regardless of age class) were equally at risk from discarded plastic waste. Although quantified at a lower level, vessel-related propeller strike injuries and incidental captures in recreational fishing activities were also identified.

Overall, anthropogenic impacts were concentrated near highly urbanised areas of the northeast coast of the North Island, an area that is considered to overlap a core habitat for green turtles in New Zealand (Chapter 1, Chapter 5). Although bycatch in commercial fisheries was not investigated as part of this work, recent work (Godoy 2016) suggests green turtles are also at risk in this northeast region, therefore potentially compounding the impact on this species. In addition, given that green turtles in New Zealand derive from widely dispersed discrete management units from across the Pacific (Chapter 3), the impacts identified herein when undertaking threat assessments and developing appropriate management measures should be considered for a number of regional genetic stocks.

6.2 Future directions

This research has investigated several key aspects of the green turtle in New Zealand waters in order to provide a baseline understanding of this species at the edge of their range in the southwestern Pacific. The results and conclusions herein provide an opportunity for future research to build upon and therefore several areas of further enquiry are suggested. The results of Chapter 2 show an exponential increase in the number of documented records over time, thus, this aspect warrants further investigation to identify the key drivers of this increase. Certainly, increased awareness during the course of this study has likely influenced the increase in reported observations. Despite this, investigating other potential environmental drivers would be valuable given the increased poleward extension of the East Australian current (EAC) over recent decades. Therefore, given that the New Zealand green turtle aggregation delineates the southern range limit of this species in the Pacific, monitoring their presence in a more systematic manner would provide a valuable opportunity to investigate how this species responds to climate-mediated temperature shifts. Investigating their response and niche expansion may also have wider application in to how other temperature constrained species may respond in light of rising global temperatures.

Chapter 3 describes the regional connectivity of the aggregation in New Zealand, yet it was considered that the sample size was not adequate to undertake mixed stock analyses and provide statistically valid confidence limits around estimates of proportional contribution (FitzSimmons et al. 1999). Therefore, further sampling would augment the genetic data collected to date and enable an assessment to determine the proportional stock composition of each contributing rookery in this foraging aggregation.

In Chapter 5, it was noted that it was unclear whether the synthetic debris items ingested by green turtles in New Zealand proportionally reflect the quantity of marine debris discharged or they are selectivity consumed soft white and clear/translucent plastics over other items. Therefore, undertaking a parallel study into the ingestion of plastic debris and shoreline surveys to quantify the availability of synthetic debris would enable this aspect to be examined. In addition, results could be compared to other areas where similar studies have been undertaken (e.g. Schuyler et al. 2012). Although outside the scope of this study, further investigation into the absorption and sub-lethal impacts of organic contaminants as a result of plastic ingestion would be beneficial. Histopathological assessments of fresh carcasses during post-mortems would also be valuable in examining aspects such as parasite loads and determining more conclusively the likely causes of mortality in stranded or incidentally captured turtles.

An overarching feature of this research is that much of the data were obtained from incidentally captured and stranded turtles. Although it has provided some important insights not previously established for this aggregation, there are some limitations which could be addressed by undertaking further research into the biology of free-ranging green turtles in New Zealand. In this manner, several of the conclusions posited as part of this work can be tested. For example, although the dietary study herein provides a description of the diet (Chapter 4), it only constitutes a snapshot of the diet and may not accurately reflect the foraging ecology of a healthy free-ranging population. By undertaking further dietary investigations in a free-ranging population would enable direct comparisons to be made with results obtained to date. In addition, quantifying food availability at local habitat sites occupied by resident green turtles can be used to investigate aspects of diet selection. Augmenting the data collected to date in this way will also enable an investigation into the nutritional constituents (energy and macronutrient content) of green turtles in New Zealand. For instance, a nutritional analysis of the diet in free-ranging green turtles, as well as the available food items, may elucidate aspects of diet selection, and determine if green turtles in this temperate habitat are satisfying their nutritional requirements. Health assessments of free-ranging turtles would also provide baseline body condition indices and normal health parameters.

Prior to this body of work, very little consideration had been given to the green turtle in New Zealand waters. In 2007, in response to inadequate standard operating procedures for the handling and recovery of stranded marine turtles by the Department of Conservation in Northland, a report was commissioned to improve the recovery of live stranded turtles (Godoy 2007; Appendix 1). With a parallel effort to increase public awareness around marine turtle presence in Northland, the improved stranding response procedures led to a notable increase in the number of reports and recoveries of live stranded turtles. Despite progress made to date, several gaps in our understanding and management of marine turtles in New Zealand remain. Therefore, given that this work constitutes new information about this globally endangered species in New Zealand, the Department of Conservation should review this current body of research and readdress the management of this species in New Zealand.

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Appendix 1: Table S1. *Chelonia mydas* mtDNA control region haplotype frequencies across Pacific breeding stocks and regiona foraging grounds based on ca. 770 bp. Sample size (*n*) for each sample location is shown.

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
Haplotype	**n**	**81**	**102**	**97**	**64**	**31**	**128**	**538**	**36**	**48**	**17**	**77**	**120**	**20**	**126**	**125**	**178**	**169**	**161**	**91**	**263**	**164**	**788**	**42**
CmP47.1		2	93	66	9	1	-	-	-	-	1	-	-	-	-	100	145	127	50	21	30	45	-	18
CmP80.1		3	8	21	14	-	-	-	-	-	-	-	-	-	-	7	15	23	13	9	7	32	-	3
CmP44.2		-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	4	-	1
CmP85.1		-	-	-	18	-	-	-	1	-	-	-	-	-	-	5	2	2	-	-	-	23	-	1
CmP20.1		-	-	-	-	3	91	276	29	47	-	-	-	-	-	-	1	2	3	4	10	3	1	1
CmP22.1		-	-	-	-	-	22	1	-	-	10	-	-	-	-	1	-	-	-	-	1	2	-	2
CmP24.1		-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	1
CmP4.1		-	-	-	-	-	-	-	-	-	-	13	58	2	26	-	-	-	-	-	-	-	1	2
CmP4.3		-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-
CmP4.2		-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-
CmP4.4		-	-	-	-	-	-	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	-	1
CmP4.5		-	-	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	-	-	-	-	-
CmP4.6		-	-	-	-	-	-	-	-	-	-	-	7	7	51	-	-	-	-	-	-	-	-	4
CmP4.7		-	-	-	-	-	-	-	-	-	-	-	-	-	31	-	-	-	-	-	-	-	-	4
CmP4.8		-	-	-	-	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-	-

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
Haplotype	**n**	**81**	**102**	**97**	**64**	**31**	**128**	**538**	**36**	**48**	**17**	**77**	**120**	**20**	**126**	**125**	**178**	**169**	**161**	**91**	**263**	**164**	**788**	**42**
CmP4.9		-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	1
CmP4.11		-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-
CmP4.14		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1
CmP4.17		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1
CmP.80.4		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1
CmP44.1		46	-	-	11	-	-	-	-	-	-	-	-	-	-	2	3	4	38	29	124	19	-	-
CmP98.1		8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	2	19	8	25	3	-	-
CmP81.1		7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	7	3	9	-	-	-
CmP171.1		1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
CmP108.1		2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP68.1		2	1	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	6	-	10	-	-	-
CmP84.1		4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	1	5	-	-	-
CmP44.4		1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP6.1		-	-	-	-	-	-	-	-	-	-	40	-	-	-	-	-	-	-	-	-	-	1	-
CmP44.5		1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP134.1		1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
CmP169.1		1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	-	-	-
CmP170.1		2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	3	2	-	-	-
CmP180.1		-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
CmP193.1		-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-
CmP168.1		-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP164.1		-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-
CmP83.1		-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	1	5	-	-	-
CmP49.1		-	-	1	1	-	-	-	-	1	-	-	-	-	-	-	-	-	2	1	5	3	-	-
CmP47.5		-	-	1	1	-	-	-	-	-	-	-	-	-	-	1	1	-	-	-	-	-	-	-

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
Haplotype	**n**	**81**	**102**	**97**	**64**	**31**	**128**	**538**	**36**	**48**	**17**	**77**	**120**	**20**	**126**	**125**	**178**	**169**	**161**	**91**	**263**	**164**	**788**	**42**
CmP173.1		-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-
CmP117.1		-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-
CmP160.1		-	-	-	1	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	1	1	-	-
CmP31.1		-	-	2	2	-	-	-	-	-	-	-	-	-	-	1	1	1	-	-	-	-	-	-
CmP174.1		-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	6	-	-
CmP118.1		-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP91.1		-	-	-	-	21	-	1	-	-	-	-	-	-	-	-	-	-	-	-	3	5	-	-
CmP89.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP191.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-
CmP19.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
CmP47.2		-	-	-	-	5	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-
CmP47.3		-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-
CmP65.1		-	-	-	-	-	1	-	-	-	4	-	-	-	-	-	-	-	-	-	-	7	-	-
CmP32.1		-	-	-	-	-	12	84	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
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CmP77.1		-	-	-	-	-	1	68	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-
CmP20.2		-	-	-	-	-	-	2	6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP20.3		-	-	-	-	-	-	10	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CmP20.4		-	-	-	-	-	-	16	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
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CmP1.1		-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	553	-

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
Haplotype	**n**	**81**	**102**	**97**	**64**	**31**	**128**	**538**	**36**	**48**	**17**	**77**	**120**	**20**	**126**	**125**	**178**	**169**	**161**	**91**	**263**	**164**	**788**	**42**
CmP2.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	98	-
CmP3.1		-	-	-	-	-	-	-	-	-	-	8	-	-	-	-	-	-	-	-	-	-	134	-
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CmP7.1		-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	-	-	-	-
CmP8.1		-	-	-	-	-	-	-	-	-	-	-	1	3	-	-	-	-	-	-	-	-	-	-
CmP8.2		-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-
CmP8.3		-	-	-	-	-	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-
CmP9.1		-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-
CmP11.1		-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-
CmP12.1		-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-
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CmP15.1		-	-	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	-	-
CmP17.1		-	-	-	-	-	-	-	-	-	-	-	-	-	4	-	-	-	-	-	-	-	-	-
CmP17.2		-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-
CmP93.1		-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-
CmP57.2		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-
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CmP82.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP87.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP199.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP206.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
Haplotype	**n**	**81**	**102**	**97**	**64**	**31**	**128**	**538**	**36**	**48**	**17**	**77**	**120**	**20**	**126**	**125**	**178**	**169**	**161**	**91**	**263**	**164**	**788**	**42**
CmP22.2		-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-
CmP192.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-
CmP119.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP186.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	3	-	-	-
CmP55.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	1	3	3	-	-
CmP98.3		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1	-	-	-
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CmP159.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-
CmP200.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-
CmP184.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1	-	-	-
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CmP183.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
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CmP182.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
CmP172.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-

		Breeding Stock (*n* = 14)														Foraging Ground (*n* = 8)								
	Location	**nGBR**	**sGBR**	**Coral Sea**	**New Caledonia**	**Vanuatu**	**Marshall Islands**	**Micronesia**	**Palau**	**Guam/CNMI**	**American Samoa**	**Revillagigedo**	**Michoacan**	**Costa Rica**	**Galapagos**	Moreton Bay (MB)	Shoalwater Bay (SB)	Edgecombe Bay (EB)	Howick Group (HG)	Clack Reef (CR)	Torres Strait (TS)	Grand Sud Lagon (GLS)	Hawaii (HW)	New Zealand (NZ)
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CmP194.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-
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CmP85.2		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-
CmP97.1		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-

Appendix 2

Handling, Recovery, and Emergency First-Response for Marine Turtles

Report for the Department of Conservation,
Northland Conservancy: SAF 2007/25

June 2007

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Abstract

Five of the world's seven species of marine turtle have been recorded within New Zealand's territorial waters. Periodically, marine turtles are found stranded on the coast, found compromised and floating on the sea's surface, caught incidentally as by-catch in fisheries activities, and/or physically injured from watercraft strikes. In accordance with the Wildlife Act 1953 (Section 3) the Department of Conservation (DOC) are responsible for their protection and management. However, a recent review of standard operational procedures for the recovery of compromised marine turtles has shown there are gaps in detail and consistency for the handling, recovery, emergency first-response, and recording and reporting of incidences. Consequently, overseas literature, historical New Zealand documentation and practices have been reviewed to develop a set of recommended standard procedures suitable for New Zealand wildlife managers. This report is intended as a guideline for Department of Conservation field staff in the handling and recovery of compromised marine turtles so as to increase the chances of survival. Recommendations are made where further review is required.

Keywords: marine turtle, New Zealand, Department of Conservation, handling, recovery, first-response, guideline

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Section 1.0 Overview

Section 2.0 Emergency First-Response

Section 1.0 Overview

1.1 Introduction

Classified into two families, *Cheloniidae* (hard-shelled) and *Dermochelyidae* (leathery-shelled), there are seven extant species of marine turtle found worldwide; i.e. the leatherback (*Dermochelys coriacea*), green (*Chelonia mydas*), loggerhead (*Caretta caretta*), hawksbill (*Eretmochelys imbricata*), olive ridley (*Lepidochelys olivacea*), kemp's ridley (*Lepidochelys kempii*), and flatback (*Natator depressus*). Marine turtles range primarily within warm tropical and sub-tropical regions, with the leatherback being the only species known to actively venture into considerably higher latitudes (Paladino et al. 1990; Carriol & Vader 2002; Southwood et al. 2005).

Although New Zealand's temperate climate is considered outside their normal range, all but two species i.e. the kemp's ridley and flatback (which have restricted ranges); have been recorded within New Zealand's territorial waters (Gill 1997). It is unknown whether these individuals have arrived in New Zealand after becoming disoriented during migration, passively dispersed outside their natural range, actively blown off course by strong weather events, or whether they comprise part of a naturally occurring population (at least seasonally) around New Zealand. As yet no detailed study has been undertaken.

Marine turtles are most commonly observed around northern New Zealand (Gill 1997; pers. obs.), however leatherbacks are often observed farther south with sightings recorded as far as Oamaru (approx. 45°S) (DOC Herpetofauna Database) and the Foveaux Strait (Eggleston 1971). Sightings are most frequent during the warmer summer-autumn months when sea surface temperatures are highest (Gill 1997). The two most commonly encountered species are green and Leatherback turtles (Gill 1997; pers. obs.).

Marine turtles are slow-growing and long-lived, sexual maturity occurs late in life, and recruitment is low (Kemf et al. 2000). Accordingly, these species are vulnerable to anthropogenic disturbance, with human exploitation, trade, and habitat destruction all contributing to significant declines in the abundance of each species; some even to localised extinction (SPREP 1989). Six of the seven species are now recognised as threatened, endangered, or critically endangered under the World Conservation Union (IUCN) 1996 Red List of Threatened Animals. Human interference affects every stage of their life cycle, from loss of nesting beach habitat and foraging grounds to mortalities at sea through incidental catch from commercial fishing practises.

In accordance with section 3 of the Wildlife Act (1953) it is the responsibility of the Department of Conservation to protect and manage marine turtles in New Zealand's territorial waters. Periodically, marine turtles are found stranded, injured, or caught incidentally as by-catch in fisheries activities. Most commonly, marine turtles recovered in New Zealand are found stranded and suffering hypothermic symptoms.

Since records began in 1837, the number of reports of marine reptiles has "increased exponentially with time" (Gill 1997). Whether this is purely a reflection of increased observer activity or a true estimation of population trend, the number of marine turtles found alive and in need of emergency care will continue. Consequently, this advice document is intended for field staff to allow an understanding on standard emergency first-response procedures when attending an incident where a marine turtle has sustained trauma. Recommendations are also made for further review and development.

1.2 External morphology and species identification

Understanding the basic morphological structures of marine turtles enables accurate identification of species and, helps describe the nature of injuries and abnormalities encountered when attending a marine turtle incident. In cheloniids (hard-shelled turtles) the main characteristics used are the structure and arrangement of the scales (scutes) of the carapace (top shell) and the presence/absence and number of prefrontal scales between the eyes (Fig. 1b,c). Demochelyids (represented only by the leatherback) do not exhibit distinctive head scales in adults (Wyneken 2001) yet can be distinguished from other marine turtle species by it's unique carapace i.e. the shell is covered with a smooth leathery skin punctuated by 7 ridges, 5 dorsally and 2 marginally (Fig. 2).

Although size, colour, shape of the jaw, skull, and body are somewhat secondary characteristics, all should be taken into consideration when distinguishing species and describing an individual. The basic external structures of marine turtles are outlined below (Fig. 1) for use in the identification key (Fig. 2).

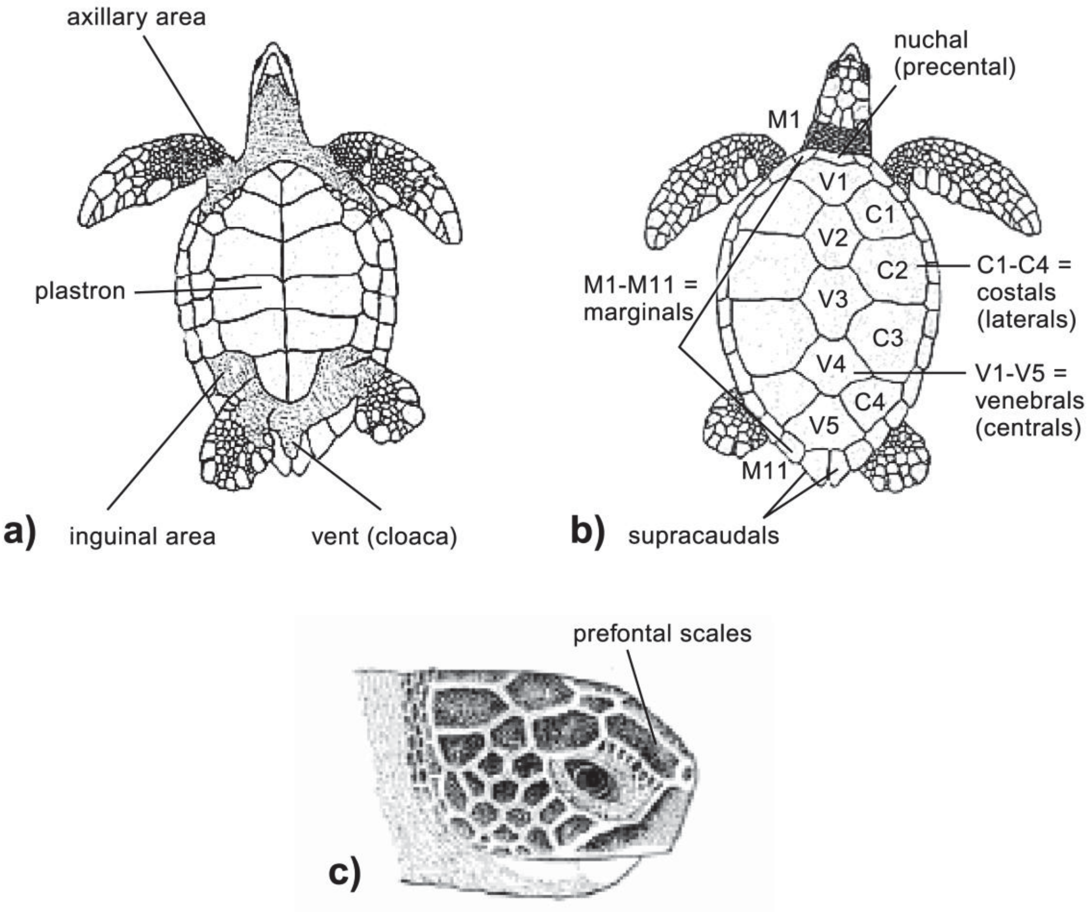


Figure 1. Ventral (a) and dorsal (b) view showing basic external structures of a Cheloniid marine turtle. Figure 1c shows close up side-view of a green turtle head, showing prefrontal scales, (source: Pritchard & Mortimer 1999).

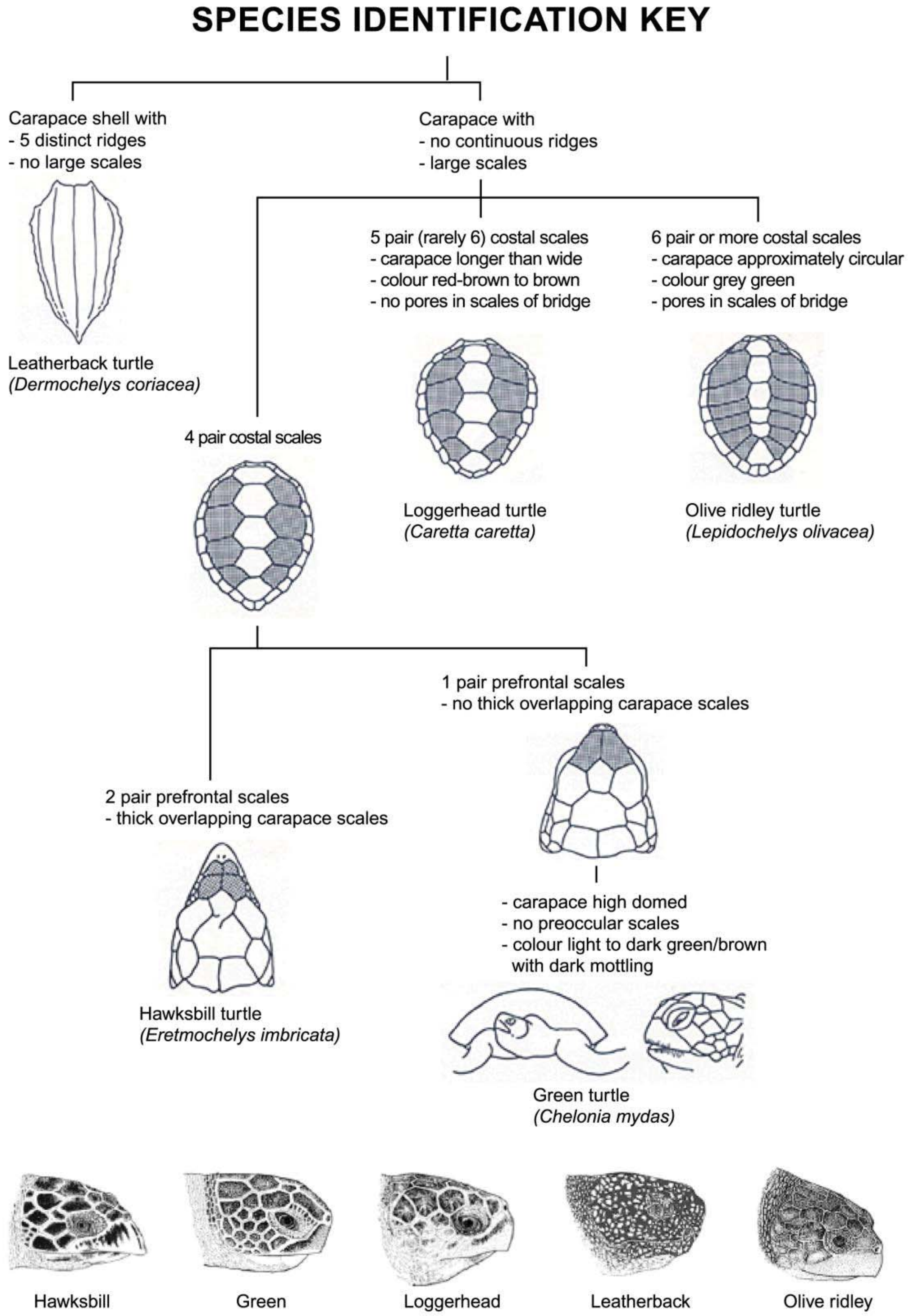


Figure 2. Species identification key (modified with permission from: Indo-Pacific marine turtles identification key; *Queensland Department of Environment and Heritage*) and close up side-view of head for each species, (source: Pritchard & Mortimer 1999).

1.3 Handling marine turtles

Care should be taken when handling turtles, not only for human safety but also to avoid injuring the animal. The best way to carry or hold a marine turtle is by firmly gripping the carapace (Fig. 3). NEVER attempt to lift a marine turtle by any of the flippers as this can cause serious damage to the limbs and even dislocate the joint.

Gently secure one hand at the anterior edge of the carapace (nuchal scale area: Fig. 1b), taking a firm grip of the area by pushing your fingers slightly under the carapace, and, placing the other hand at the posterior edge of the carapace, taking hold of the supracaudal scales (Fig. 1b). For large turtles too heavy to lift by one person, use the same method with two people or place onto a mat or in a box and carry.

Figure 3. Recommended method for holding and carrying a marine turtle (photo courtesy D. Godoy, 2007).

1.4 Measuring marine turtles

It is important to take accurate measurements of marine turtles, as it is often the only way to estimate age and maturity (see section 1.5). Standard measurements recorded from marine turtles are Curved Carapace Length (CCL) and Curved Carapace Width (CCW). CCL and CCW measurements should always be recorded on a Marine Turtle Event Data Sheet (Appendix A). Additionally, tail length can be recorded to predict sex in mature adults (see section 1.5). For CCL, CCW, and tail length, a flexible measuring tape (e.g. fibreglass) should be used (see Appendix B: Marine turtle field kit).

Curved Carapace Length (CCL): This measurement is taken ‘over-the-curve’ of the vertebral midline of the carapace (Fig. 4a), from the junction between the skin at the base of the neck and the nuchal scale (anterior end) posteriorily to the junction of the supracaudal scales (Fig. 4b).

Curved Carapace Width (CCW): This measurement is taken over-the-curve of the carapace at its widest point. Place the tape along a perpendicular mid-line axis and measure between the two most extreme marginal scutes (Fig. 4c).

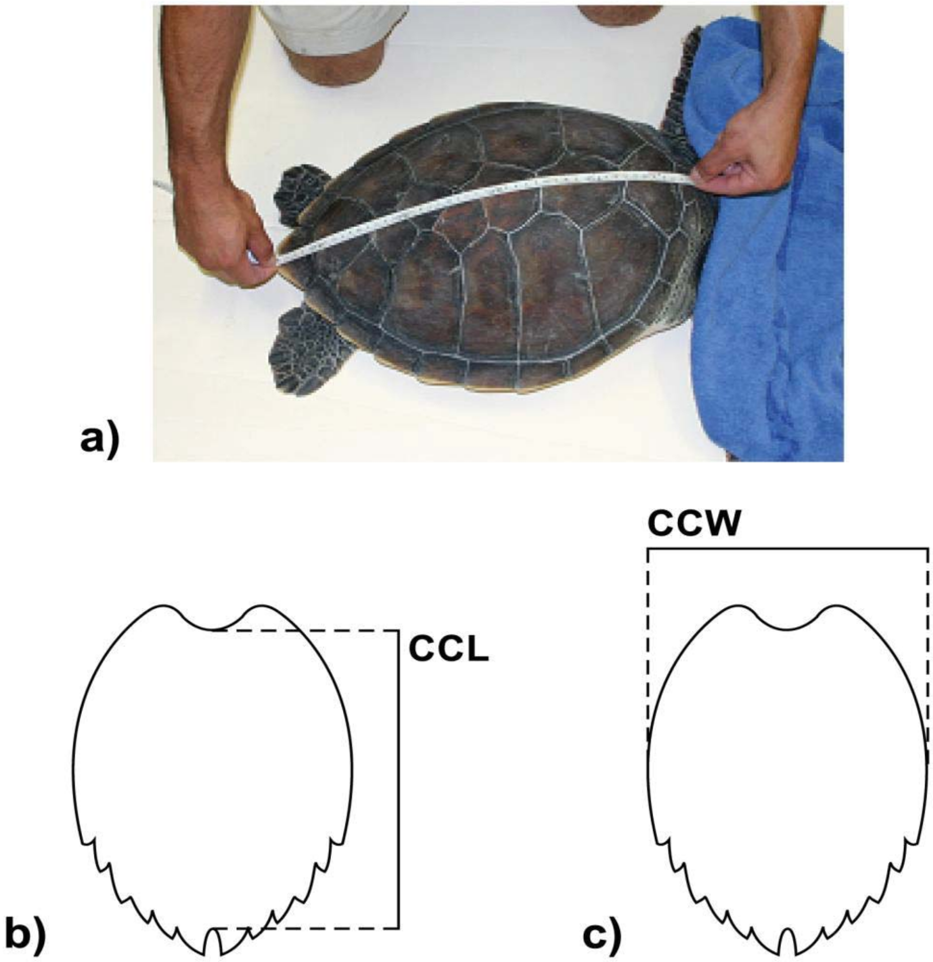


Figure 4. Measurements are taken ‘over-the-curve’ (a) of the carapace for Curved Carapace Length CCL (b) and Curved Carapace Width CCW (c). Source: a (photo courtesy of D. Godoy, 2005); b, c (Bolten 1999).

1.5 Sex determination

Marine turtles do not exhibit external reproductive structures, and secondary sexual characteristics only develop in mature adults (Wyneken 2001; Phelan & Eckert 2006). Morphometric differences that can help determine sex in mature adults are carapace morphology, elongation and curvature of the claws on the fore-flippers of males, and the most conclusive characteristic being tail length in mature males.

It should be noted that marine turtles do not reach sexual maturity until reaching a carapace length of >60 cm CCL. With the exception of the leatherback, most marine turtles encountered in New Zealand fall under this size class (Gill 1997; pers. obs.), hence it is unlikely sex determination will be achieved through external examination (Bolten 1999; Wibbels 1999; Chaloupka & Limpus 2001). Nevertheless, methods to measure these differences have been included in the event it is encountered (Fig. 5).

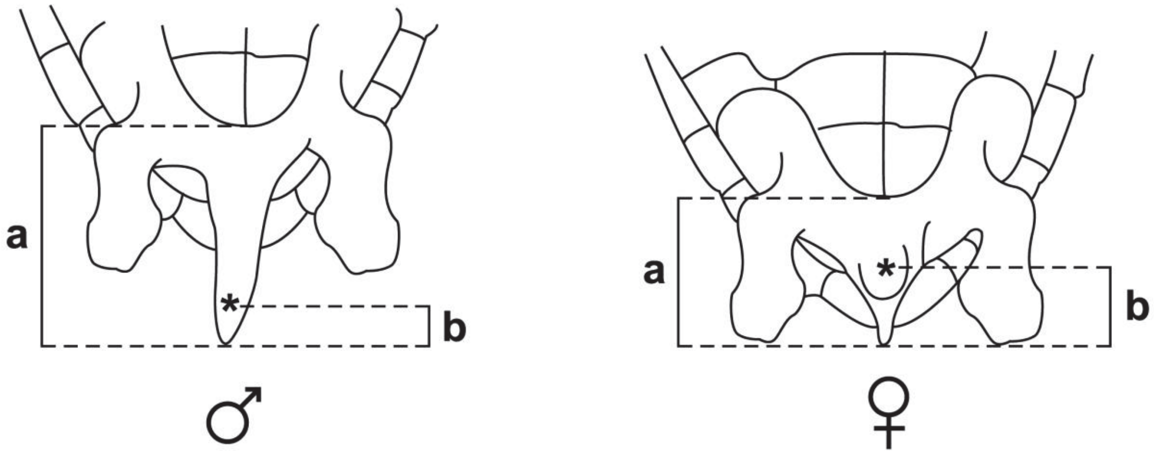


Figure 5. Two tail length measurements where a = Total Tail Length (TTL), taken from the posterior margin of the plastron to tail tip and; b = Post-cloacal Tail Length (PTL), taken from the vent (cloacal opening) to tail tip (source: Bolten 1999). Relative differences between mature male (♂) and mature female (♀) are shown.

1.6 Record keeping

The Department of Conservation maintains the *Herpetofauna Database* and is responsible for the recording and administration of data relating to marine turtles in New Zealand's territorial waters. For each event, a separate Marine Turtle Event Data Sheet (Appendix A) should be filled out, and it is important that **ALL** marine turtle sightings and encounters are reported to:

Herpetofauna Database Administrator
Department of Conservation
PO Box 10-420
Wellington

N.B. It is recommended current DOC reporting procedures, record administration, and administrator contacts are reviewed and amended to reflect current management objectives (see section 2.7).

1.7 Receiving reports

When receiving reports from members of the public it is important to collect all relevant information before making a decision on the best course of action. For example:

Any marine turtle reported found along the shoreline, even if not showing signs of physical injury or illness, is most certainly compromised and will require immediate emergency care. In these cases, the situation should be treated as a stranding event (see section 2.0).

A marine turtle that is encountered and reported at the sea's surface, although seemingly 'sunning' itself yet remaining relatively motionless even when approached, could be compromised, cold-stunned, or injured. Staff should look at responding to the event not merely as a sighting, but as a situation requiring appropriate action (see section 2.0).

1.8 Photographs

A photograph is one of the most important pieces of information you can obtain from a sighting, stranding, or other event. It is the only way to clearly identify a species without first hand examination by trained staff or researchers. Although the Species Identification Key (Fig. 2) will suffice in most instances, morphological anomalies do occur, e.g. green turtles can have 5 pairs of lateral scales instead of the normal 4 pairs, and even only on one side. Anomalies such as these may occur from hybridisation between two species e.g. mating between green and flatback turtles have been known to produce viable offspring (D. Donnelly, pers. comm.).

Even when receiving sighting reports from members of the public, be sure to ask if they have or can take images for accurate species identification. To effectively determine species, clear photographs of key characteristics of a turtle's anatomy should be taken, including the head region and scale arrangement of the carapace (Fig. 6 & section 1.2).

Photographs considered essential for correct identification include:

- A full frame lateral view.
- A full frame dorsal view showing carapace detail (Fig. 6).
- A close up frontal view of the skull showing the prefrontal scales (Fig.6).

Additional photographs can include:

- A close-up view of the side of the head showing lateral scales (Fig. 6).
- Any other important features i.e. tags, entangling debris, epibionts, and injuries.
- A close-up view of the upper surface of the fore-flippers.
- A ventral view of the plastron.

Figure 6. The photograph above (left) shows the carapace of a green turtle, however, an abnormality of the 4th vertebral scute (appearing as 3 separate scutes) can be seen. When attempting to identify a species, this may lead to confusion when a photograph is not available for review. The photograph above (right) shows the arrangement of the pre-frontal scales seen in a green turtle (photos courtesy D. Godoy, 2005).

1.9 Health and safety

Priority should be given to human health and safety. Staff members responding to any marine turtle incident should take precautions to minimise the risk of injury or infection.

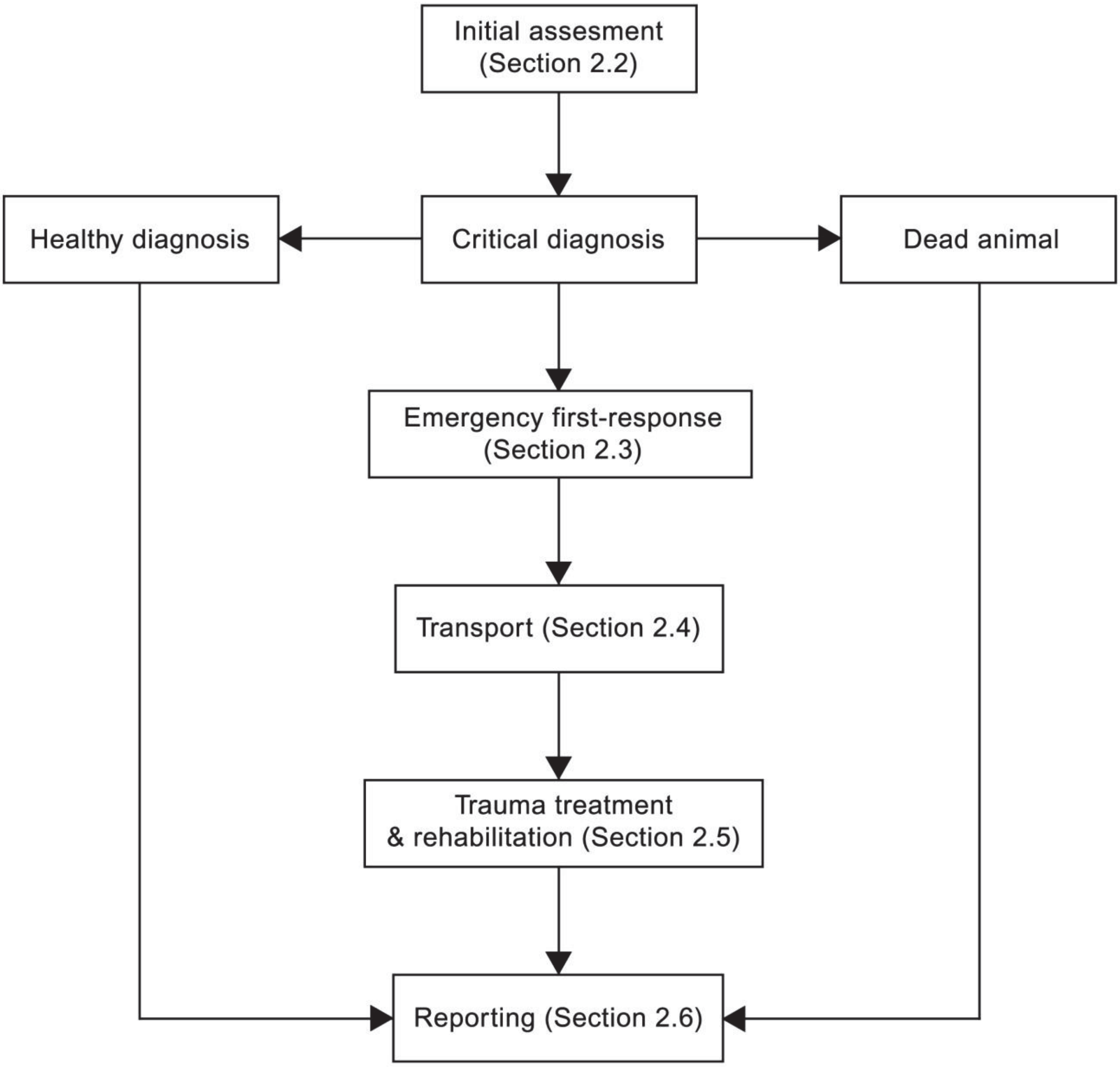
- An initial site assessment should be made to avoid injury to responding staff, members of the public, or to the marine turtle.
- Marine turtles, particularly seriously ill individuals, can harbour a number of pathogens that could potentially be infectious to humans. Hence, it is advised latex examination gloves be worn when handling marine turtles.
- When the turtle is still conscious and active, it is also recommended wearing sturdy (leather) gloves, as marine turtles can bite causing serious bites injury. Handlers should also keep their hands and fingers away from the animals' mouth.

- A turtle in good health will often thrash about when handled; therefore care should be taken to avoid being struck with the front flippers, particularly with larger turtles. Being struck with a front flipper can inflict serious bruising as well as injury from the claw(s) protruding from the leading edge of each front flipper.
- Cover surface wounds with protective dressings to avoid infection of the wound. Dressing can be fixed in place with standard first aid dressing tape.
- Any samples collected should be placed in sealed containers or bags, clearly labeled, and transported appropriately to avoid direct contact or contamination.
- Seek medical advice or treatment for any bites, cuts, or other injuries sustained during handling.
- **Hazard warning for person informing about marine turtle incident** – any member of the public reporting a stranding or similar encounter with marine turtle where they come in close contact with the animal should be informed of the potential hazards arising from such an encounter. They should be advised NOT to handle the animal unless the animal is in immediate danger and/or specifically advised by DOC staff.

Section 2.0 Emergency First-Response

2.1 Response flow chart

Staff members responding to and attending any marine turtle incident other than a 'Sighting Only' should follow the flow chart below to determine best course of action.



2.2 Initial assessment

It is important that accurate assessments are made when responding to a marine turtle incident. Often, the emergency first-response received by a marine turtle will substantially increase the animal's chance of survival.

All staff responding to and attending a marine turtle incident will need to make decisions, often in the field, as to the best course of action. To achieve this, an initial hands-on assessment of the animal should be carried out. The initial assessment is based on determining if there are any obvious signs of **immediate danger, injury** and the level of **responsiveness**. Use section 1.2 (External morphology and species identification) and the guide below to determine and describe the condition and health status of the animal. Remember to take with you a suitable field kit (see Appendix B) and record all events on a Marine Turtle Event Data Sheet (Appendix A).

Status: Healthy

- The turtle is not in any immediate danger.
- The turtle has NOT been found stranded on the shoreline (see grey box below).
- It shows no signs of external injuries or deformities.
- The carapace is not heavily covered with epibiont growth.
- The turtle becomes active when approached; lifting it's head strongly, and moving its flippers in an attempt to swim.
- It is not entangled in any foreign debris, or shows no signs of having ingested a hook or other foreign object.

⇨ **Healthy diagnosis**

- Leave the animal as it was found.
- Record event details (Appendix A).
- Determine species (see section 1.2)

- Take photographs of the head, carapace, tags if present, and other relevant areas.
- Measure turtle (see section: 1. 4).

Status: Critical

Note: Any turtles found on the coast, even at the water's edge, should be considered sick or injured, and be treated as such.

- The turtle is in immediate danger.
- The turtle IS found on the shoreline.
- The turtle shows signs of physical injury or deformity.
- Is behaving erratically in an uncontrolled manner.
- The turtle is entangled or has ingested foreign debris.
- The carapace is heavily covered with epibiont growth.
- Note: unless required for emergency treatment of an injury or obstruction, there is no need to remove epibiotic growth from the carapace. If epibiotic growth is to be removed, one can lightly use a scraper, brush, or cloth with water.
- The turtle does not become active or move when approached; it does not lift its head or try to move away in a controlled manner.
- If it is lifted, it does not respond by lifting its head or display swimming/crawling motion with its flippers.
- The turtle is inactive, showing no response even when lightly touched on or around the eye.
- There is no attempt to retract the flipper when lightly pulled or squeezed.
- The turtle has been assessed for hypothermia (section 2.3.1) and been given a severity Class 2, 3, or 4.

⇨ **Critical diagnosis - Go to Section 2.3: Emergency first-response.**

Status: Dead

- The turtle does not respond to any physical stimulus.
- The turtle shows signs decomposition (including foul smell) or *rigor mortis* has set in.

⇨ **Death**

- Record event details on a Marine Turtle Event data Sheet (Appendix A).
- Determine species (see section 1.2).
- Take photographs of head, carapace, tags (if present), and other relevant areas.
- Collect entangling debris for identification.
- Measure turtle (see section 1.4).
- Collect tissue samples if required (Appendix C).
- Recover turtle for necropsy or dispose of appropriately.

Note: Any turtles recovered for necropsy should be refrigerated and transported to the appropriate person(s) as soon as possible. A copy of the necropsy report should be supplied to the *Herpetofauna Database Administrator*.

2.3 Emergency first-response

This section is intended as an emergency first-response guide for field staff on how to address the types of trauma commonly found in compromised marine turtles in New Zealand. This guide is **NOT** intended as a medical manual for the treatment of marine turtles (see section 2.7 for recommendations), however some points have been included (grey boxes) that will assist veterinarians treating trauma in the first instance until expert advice can be sought (see section 2.5 & Appendix D).

The emergency treatment required for marine turtles is much the same as for any other injured or compromised vertebrate however, ectothermy in marine turtles requires additional considerations in the New Zealand context, namely assessing and treating for hypothermia or ‘cold-shock’. The primary goal for field staff should be to:

- Remove the turtle from any immediate danger.
- Attend to and minimise critical trauma.
- Stabilise the animal.
- Transport the animal to an emergency treatment facility.

Although the sections below are separated into common types of trauma and the treatment for each, when marine turtles are recovered in need of emergency care, they will often exhibit symptoms consistent with more than one type of trauma. Therefore, it is recommended to familiarise staff with the correct treatment procedures for all types of trauma, to ensure proficiency in determining best course of action (see section 2.7). It recommended a standard systematic approach to assessing the condition of a marine turtle should be followed. Some important considerations include: general body condition, skin condition (e.g. presence of superficial lesions), overall musculature and fat deposition, degree of alertness, responsiveness, and strength, head and body symmetry, ocular abnormalities (e.g. discharge, sunken or swollen eyes), nasal discharge, respiratory difficulties, epibiont growth.

Note: Treating symptoms quickly in the first instance will greatly increase the chance of survival and successful rehabilitation to full health.

2.3.1 Hypothermia

For marine turtles, a core body temperature between 20-30°C is considered normal (Milton & Lutz 2003). To maintain a core body temperature within this range, marine turtles rely on behaviour and environmental heat (Norton 2005). This characteristic, known as ectothermy, is common to all reptiles. With the exception of the leatherback, sudden drops or prolonged periods in water temperature below 15°C, can seriously impact negatively on a turtle’s physiology and behaviour. In these cases, their core body temperature cannot be sustained and hypothermia can ensue.

In New Zealand, seasonal sea temperatures fall well below this range. Hence, hypothermia and related secondary infections (e.g. bacterial pneumonias) is the most common type of trauma likely to be encountered by staff attending marine turtle strandings. Even if hypothermia or ‘cold-shock’ does not appear to be the primary cause of distress, turtles found compromised during the cooler winter-spring months should be suspected as suffering from hypothermia, and assessed accordingly.

The symptoms exhibited by hypothermic marine turtles (whether directly or indirectly caused) include: unresponsiveness to external stimuli, lethargy, emaciation, dehydration, carapacial and plastron lesions, buoyancy problems, and exhibit heavy epibiotic growth (e.g. green algae and barnacles) on the carapace.

Note: As a turtle begins to metabolically shut down, its ability to move, swim, and lift its head to breathe, is inhibited. Consequently, if a marine turtle is found hypothermic either at sea or washed ashore, it is likely it has ingested and possibly inhaled seawater and should be placed into the resuscitating position (see section 2.3.7).

Figure 7. This photo shows the lethargy (unresponsiveness) typically exhibited in hypothermic marine turtles. Note: upon presentation, this juvenile loggerhead was unable to lift its head or flippers or respond to external stimuli. This turtle made a full recovery within three weeks (photo courtesy D. Godoy, 2004).

Assessing hypothermia

The first step in determining the severity of hypothermia is to determine the level of responsiveness of the turtle to external stimuli (Sadove & Pisciotta 1998), i.e. a healthy turtle will respond strongly whereas a compromised turtle will be lethargic and largely unresponsive. This simple test is based on the response reflexes exhibited when different parts of the animal's body are stimulated. The test includes:

- Head lift response: When a turtle breathes it should lift its head above the horizontal plane.
- Eye reflex: lightly touch the eyelid and observe any response.
- Head reflex: lightly touch the nares and observe any response.
- Limb reflex: lightly squeeze a flipper and observe any response.

- Tail reflex: lightly touch the cloaca (tail area) and observe any response.

This assessment enables field staff to classify the level of severity presented into one of four classes. The Class score (1-4) should be recorded (with comments) on the Marine Turtle Event Data Sheet (Appendix A).

Class 1: Response reflexes 1-5 (listed above) above are strong.

- The turtle is active and exhibits strongly controlled movements of its flippers in an attempt to swim or crawl when approached.
- The head is lifted easily above the horizontal plane every time it takes a breath.
- There is a strong response when the eyes, nares, flippers, or tail is lightly touched.
- The turtle reacts strongly when handled and attempts to swim vigorously when picked up.

Class 2: Response reflexes 1-5 (listed above) above are moderately strong.

- There is a moderate reaction when the turtle is approached showing some attempts to swim/crawl, however may appear to be slow or uncontrolled.
- Head lift is above the horizontal plane but is weak or incomplete.
- There is some limited reaction when the eyes, nares, flippers, and tail are stimulated, however is relatively weak and slow.
- The turtle reacts weakly, slowly, or uncontrolled when handled and no attempt is made to swim when picked up.

Class 3: Response reflexes 1-5 (listed above) above are weak.

- There is no attempt to move when approached, or any movement is minimal, weak, or uncontrolled.
- Head lift when breathing is absent or is very weak, or does not go above the horizontal plane.

- Eye, head, flipper, tail response is absent or slow and very weak when stimulated.
- The turtle is limp when handled and no attempts to move are made when picked up.

Class 4: Response reflexes 1-5 (listed above) above are absent.

- There is no movement when approached, even when animal is lifted.
- Head lift is absent or breathing is not observed.
- There is no response when the eye, nares, flipper, or tail are stimulated.
- The turtle is limp when handled and no attempts to move or swim are made when picked up.

Taking internal body temperature

It is recommended to record as soon as practicable, both internal body temperature and the temperature of the surrounding seawater from where the turtle was recovered. It is important to ensure that body temperature is **NOT** raised too quickly (see veterinary notes below in grey box) during initial treatment. A thermometer with a range between 0-35° C is recommended for taking cloacal temperatures (see Appendix B for field equipment list). When taking internal body temperatures from marine turtles, the easiest and most accurate location is from the cloaca. Carefully lift the rear of the turtle and insert a lubricated (e.g. using *K-Y Jelly*) thermometer into the cloaca until the probe is inserted approximately 2.5cm. Hold in place until reading stabilises and record.

Figure 8. Careful insertion of a lubricated thermometer approximately 1 inch should be sufficient.

Field response

- Externally examine and assess the degree of responsiveness.
- Take a deep cloacal temperature and environmental temperature of the surrounding water. Notify veterinarian of temperatures upon presentation.
- Place turtle in resuscitation position (see section 2.3.7) and carefully place the turtle into a padded carry box (see section 2.4).
- Place a towel or mat over the turtle's carapace.
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

Veterinary notes for treating hypothermia.

Hypothermic turtles should be treated carefully as a sudden rise in core body temperature may cause sudden shock and death.

Emergency treatment:

- Maintain in resuscitation position (see section 2.3.7) until satisfied the turtle is breathing adequately, and there are no signs water has been inhaled.
- **Slowly** raise the core body temperature by no more than 0.5° C/hr, and no more than 3-4° C in the first 24 hours. Continue to raise body temperature at this rate until an internal body of 18° C is reached (cloacal temperature can be taken).
- Administer fluids (e.g. Hartman's solution) subcutaneously at a rate of 10-15 ml/kg/day for the first 24 hours. Do not exceed 3% of total body weight per day.
  - Fluids can be administered into the inguinal area anterior to the hind limbs or into the axillary area anterior to the foreflippers.
- **Do not** attempt to give food or antibiotics until their core body temperature has risen and stabilised.
- If too weak to lift its own head, **DO NOT** place in water deeper than it's nares (nostrils). Preferably, if it is weak, keep out of water or first 24 hours, but keep covered with damp towel and at an ambient temperature that will not cool it.
- If the turtle is strong enough to swim and lift it's own head to breathe, then it can be placed in a fresh- or brackish-water bath (avoid seawater initially) at no more than 4° C higher than the water temperature from where it was recovered.
- Avoid using dry air heaters unless adequate hydration of the turtle can be maintained.
- Keep eyes moist using standard animal eye ointment.
- After 24 hours administer 5% glucose solution subcutaneously.

2.3.2 Watercraft strike

Injuries sustained from watercraft strikes are usually serious, and in many cases fatal (Fig. 9). Emergency medical treatment should be sought even when external signs of trauma are minimal. This is important because these incidents often cause severe internal injuries which are not immediately discernable, yet can develop into a host of other secondary infections and trauma. If a marine turtle has carried an injury for some time, it may have stopped feeding and also show signs of hypothermia, drowning, emaciation and dehydration, buoyancy problems, which should be addressed.

Figure 9. The injuries shown on the carapace of this decomposing juvenile green turtle are consistent with those observed in marine turtles following a watercraft strike (photo courtesy D. Godoy, 2007).

Field response

- Externally examine and assess the degree of responsiveness.
- Gently cleanse any wounds with saline solution and remove all surface debris.
- Flush with a 1:10 dilution of povidone-iodine solution.
- Control heavy bleeding by direct pressure or packing the wound with clean gauze or towels. Apply antibiotic ointment (e.g., triple antibiotic), bandage and tape dressing in place.

- Stabilise the turtle so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.3 Entanglement

Sea turtles can become entangled in discarded fishing gear or rubbish (Fig. 10). Entanglement by fishing line (monofilament) can cause severe lacerations and constrict blood flow. Any marine turtle found entangled (unless clearly uninjured and healthy), should be assessed by a veterinarian. Entanglement can lead to a number of other issues that will need emergency care. If a turtle has been entangled underwater for a prolonged period, even though it may not show signs of drowning, should be handled as if it may have drowning symptoms as a precaution (see section 2.3.7). If a turtle has been entangled for a prolonged period, it may have been unable to feed or breathe effectively, hence be suffering from other secondary symptoms (e.g. dehydration, emaciation) and infections (e.g. septicaemia).

Field response

- Externally examine and assess the degree of responsiveness.
- Gently remove or cut away any loose monofilament or other debris and retain for identification.
- Any tightly bound debris or monofilament, unless restricting breathing should be left in place for expert medical care.
- Stabilise the turtle so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

Figure 10. This leatherback had been entangled for some time and suffered many of the symptoms described above. The photo on the right shows the infected wound caused by the rope. This animal eventually died (photo courtesy Dave Donnelly, 2006).

2.3.4 Incidental capture

Incidental capture can occur from commercial or recreational fisheries activities. Treatment will vary depending on the fishing method used. Even though turtles can survive extended periods without breathing, all animals caught in nets and long-lines should be treated for drowning. Marine turtles caught by fishing hooks can either be foul-hooked or have ingested the bait - been hooked in the throat, stomach, or intestine. If a turtles is landed hooked, and the hook cannot be readily removed, the animal should be assessed and treated by a veterinarian.

Field response

- Externally examine and assess the degree of responsiveness.
- Gently remove or cut away any loose monofilament or other debris (net gear) and retain for identification.
- Any **tightly** bound monofilament or net gear, unless restricting breathing should be left in place for expert medical care (see section 2.3.3).
- If the turtle has been caught in a net, treat for drowning (see section 2.3.7).
- If the turtle has been landed foul-hooked, and the hook is visible **do not** attempt to remove the hook unless the barb is clearly visible and can be cut. Cut the line at the head of the hook, so that it cannot entangle further. Treat for drowning.
- If the turtle has been landed and has been hooked in the throat, stomach or intestine, **do not** attempt to remove the hook or pull the line. Cut the line so that a small length remains outside the mouth but not long enough to entangle further. Without placing tension on the line, tape the line to the carapace before transport.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4) to minimise further injury.
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

If a hook needs to be removed from the throat, use a vet gag or PVC pipe to keep the mouth open safely (Fig. 11). It may also be necessary to anaesthetise the patient.

Figure 11. Method for safely maintaining mouth open in a marine turtle. A PVC (above left) or other rigid plastic (above right) bite pipe, or vet gag can be used (photos courtesy of D. Godoy, 2007).

2.3.5 Buoyancy problems

Buoyancy problems may arise from a number of causes including: pneumonia, gastrointestinal disease arising from gastrointestinal obstructions, free air accumulating in the coelomic cavity (originating from the respiratory or gastrointestinal tracts), and trauma or injury causing air to be trapped in the coelomic cavity. A marine turtle afflicted with a buoyancy disorder will normally be found floating at the seas surface, unable to dive to feed, escape predators or watercraft, and may suffer considerable sunburn. If a marine turtle has been buoyant for some time, the symptoms observed will be very similar to those of hypothermic turtles. In the case of buoyant turtles, diagnosing the primary cause is nearly impossible in the field and therefore it is recommended to follow the same treatment as for hypothermic animals until expert advice can be sought.

Field response

- Externally examine and assess the degree of responsiveness.
- Stabilise the turtle in a position so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4) into a resuscitation position for drowning (see section 2.3.7).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.6 Ingestion of foreign debris

Unless there is obvious evidence a marine turtle has ingested foreign debris, it is very hard to diagnose directly. Marine turtles that ingest foreign debris may cause a gastrointestinal obstruction leading to a compaction of the gut or intestine. In severe cases the animal will stop feeding and eventually die. The symptoms seen in turtles that have ingested debris are similar to various other traumas. Emaciation, dehydration, lethargy, carapacial and plastron lesions (due to secondary infections), buoyancy problems, and heavy epibiotic growth (e.g. green algae and barnacles) on the carapace, can all be seen in these cases. Any turtles suspected of having ingested foreign debris should be recovered for immediate expert treatment. The emergency field response is similar to that for hypothermia.

Field response

- Externally examine and assess the degree of responsiveness.
- If debris (e.g. plastic bag or monofilament) is observed hanging from the mouth, **gently** pull on it.

- If no resistance is felt, slowly remove and retain for identification.
- If resistance is felt when pulling on the ingested debris, **STOP - DO NOT PULL FURTHER**. Cut the bag, monofilament, or debris as close to the body of the turtle as possible to avoid further entanglement.
- If monofilament is observed protruding from both the mouth and the cloaca, trim as close to the body as possible to avoid further entanglement.
- Stabilise the turtle so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

In cases where a gastrointestinal obstruction is suspected, an x-ray of the body is recommended.

2.3.7 Drowning

Although marine turtles can remain submerged underwater for extended periods, most turtles if not all that are found stranded along the shoreline, incidentally caught in fisheries activity, or found compromised at sea, have most likely ingested and possibly inhaled some amount of seawater. It is important to note that marine turtles that may appear dead due to the incidents described above, may be still alive but in a comatose state. They may not respond to external stimulation of the eye, exhibit any deep pain reflexes (Norton 2005), breathe, or have a detectable heartbeat.

If staffs are unsure whether a marine turtles is dead, the animal should be placed into a resuscitation recovery position that will enable water to freely drain from the oesophagus and lungs. The easiest way to do this is to place a rolled blanket under its plastron and hind flippers so as to slightly elevate the lateral plane of the patient. Care should be taken not to elevate the body to high as to restrict breathing as it resumes. Even if it does not appear to be drowning, marine turtles found in this state should be put into a position.

Field response

- Externally examine and assess the degree of responsiveness.
- Stabilise the turtle in a resuscitation position (see above).
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4) into a resuscitation position.
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.8 Oil contamination

Although no record of oil contamination affecting marine turtles has ever been observed in New Zealand, it is necessary to include in the event it is encountered. Like all marine life, marine turtles are affected by oil pollution and spills. Ingestion (causing toxicosis), skin exposure may result in sloughing of the skin and necrosis of tissue (George 1997).

Note: there is no onsite field treatment available for oil ingestion; seek medical treatment immediately.

Field response

- Externally examine and assess the degree of responsiveness.
- Gently remove any surface oil using either:
- Plain mayonnaise, which is gentle on the eyes.
- Vegetable oils (olive or sunflower) which helps break up and remove surface oils.
- A mild dishwashing detergent in warm (**not hot**) water.
- Wipe around the mouth and eyes with clean cloths dampened with food oils.

- Rinse and repeat.
- Stabilise the turtle so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.9 Predator attacks

Occasionally marine turtles may be presented suffering from injuries inflicted by natural (e.g. sharks) and unnatural (e.g. dogs) predators. Marine turtles that strand along the coast are particularly prone to being attacked by domestic dogs. Treatment for this type of trauma is the same for watercraft strikes.

Field response

- Externally examine and assess the degree of responsiveness.
- Gently cleanse any wounds with saline solution and remove all surface debris.
- Control heavy bleeding by direct pressure or packing the wound with clean gauze or towels.
- Stabilise the turtle so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.10 Fibropapillomatosis

Fibropapillomatosis is a tumour forming disease primarily found in green turtles. To date no accounts of this disease have been recorded in New

Zealand. Hawaiian green turtles appear to be the population mostly affected. As yet the cause of this condition is unknown but a viral agent (herpesvirus) is suspected (Campbell 1996). Symptoms of this condition are exhibited as papillomatous growths ranging in size from a few millimetres to >20cm. Primarily found around the head, neck, limbs, and tail, extensive growths of these tumours can obstruct sight, feeding, and swimming ability. There is no emergency field treatment for the disease itself, but if a turtles is found compromised due to this disease, the animal should be recovered and assessed and treated by a veterinarian.

Field response

- Externally examine and assess the degree of responsiveness.
- Stabilise the turtle in a position so that it cannot injure itself if it begins to thrash about.
- Carefully place the turtle onto a mat or tarpaulin and into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

2.3.11 Emaciation and dehydration

Emaciated and dehydrated turtles show considerable reduction in the condition and mass of the soft tissue around the eyes (appearing sunken), neck and shoulder region, as well as a clearly depressed plastron. Other symptoms include dry flaky skin, lack of skin elasticity and weeping eyes.

Emaciation and dehydration are often simultaneous conditions arising from a number of different traumas. Most turtles found stranded in New Zealand will be suffering from one or both of these conditions. There is no immediate emergency care that can be carried out in the field short of administering fluids subcutaneously. Therefore treatment should be focus on stabilising the animal and seeking immediate expert emergency care.

Field response

- Externally examine and assess the degree of responsiveness.
- Stabilise the turtle in a resuscitation position and place carefully into a padded carry box (see section 2.4).
- Seek expert veterinary care immediately (see section 2.5).
- Report incident.

Veterinary notes for treating dehydration.

Often turtles that are presented in this condition in New Zealand suffer from hypothermia and should **not** be given food or antibiotics until their core body temperature has risen to at least 18° C and stabilised (see veterinary notes in grey box at the end of section 2.3.1: Hypothermia). If hypothermia is not diagnosed, yet dehydration is, then:

- Maintain in resuscitation position (see section 2.3.7) until the turtle is breathing adequately, and there are no signs water has been inhaled.
- In severely dehydrated and hypoglycemic marine turtles, a sterile 5% dextrose solution (2 mg/kg body weight) can be delivered into the coelomic cavity by passing a needle into the inguinal area (Fig. 1a) just cranial to the hind limb (Campbell 1996).
- Hartmann's solution can also be administered in the same way at a rate of 10-15 ml/kg every four hours for the first 24 hours.
- Administered fluids should not exceed 5% of total body weight per day.
- If not hypothermic, but still dehydrated, place in water (freshwater or brackish water i.e. 16 ‰) at no more than 4° C higher than core body temperature.
- If too weak to lift its own head, **DO NOT** place in water deeper than it's nares. Preferably, if it is weak, keep out of water or first 24 hours, but keep covered with damp towel and at a temperature that will not cool it.
- Avoid using dry air heaters unless adequate hydration of the turtle can be maintained.
- Keep eyes moist using standard animal eye ointment.

2.4 Transporting recovered marine turtles

Turtles should be transported in a secure travel box, ideally with a padded base (e.g. closed cell foam), and if fluid inhalation is suspected, placed into a resuscitation position as described is section 2.3.7. The turtle should be monitored periodically to ensure consciousness and that nothing is obstructing its airways. If a turtle is not suspected of suffering hypothermia and the ambient temperature is seasonally warm, then a damp towel placed over the turtle across the carapace will help reduce dehydration and overheating. **DO NOT** transport in water. Even if the ambient temperature is warm, the turtle should not be exposed to wind chill while being transported, especially if covered with a wet towel.

For a turtle suspected of hypothermia, **DO NOT** transport it where it will be exposed to wind chill – this will dramatically reduce the chance of survival. Ideally, a hypothermic animal will be transported inside the vehicle with the carapace covered in a dry blanket or towel. To reduce dehydration in turtles with this condition it is best to apply a water-based lubricant (e.g. *K-Y jelly: Johnson & Johnson*) over the entire body and avoid further dehydration. Avoid overheating the area where the turtle is being transported.

Figure 12. A typical transport box (photo courtesy D. Godoy, 2005).

2.5 Trauma treatment and rehabilitation

Any marine turtle recovered by DOC staff should be assessed and treated as soon as possible by a veterinarian. Resuscitating and carrying out any emergency care should be the first priority for an attending veterinarian, followed by stabilising the animal's core body temperature and hydration. It is recommended veterinarians who are contracted by DOC for the treatment of distressed wildlife are familiar with treating marine turtles (see section 2.7 for recommendations). As soon as feasible, the turtle should be transported appropriately to the nearest rehabilitation facility for ongoing treatment. For Northland, the closet facility is Kelly Tarlton's Antarctic Encounter and Underwater World. In the interim, emergency advice can be sought from:

Kevin Turner - veterinarian
Lynfield Veterinary Clinic
122 White Swan Rd
Mt. Roskill
Auckland
Ph. 09 626 4335
Mobile: 027 727 96560
Email: lynvet@paradise.net.nz

Dr Brett Gartrell
New Zealand Wildlife Health Centre
Massey University
Palmerston North
Ph 06 356 9099 ext 7398
Email: B.Gartrell@massey.ac.nz

Dan Godoy
Earth & Oceanic Science Research Institute
Auckland University of Technology
Private Bag 92006
Auckland 1020
Mobile: 021 742 656
Email: dgodoy@aut.ac.nz

Kelly Tarlton's Antarctic Encounter and Underwater World
Andrew Christie – Aquarium Curator
23 Tamaki Drive
Orakei
Auckland
Phone: 09 528 0603 or DDI: 09 528 1720
Mobile: 021 222 1266
Fax: 09 528 5175
Email: andrew.christie@kellytarltons.co.nz
Website: www.kellytarltons.co.nz

Note: The contacts listed above are not currently contracted to DOC, therefore it is recommended a review of the emergency treatment process be addressed (see section 2.7).

2.6 Reporting

All marine turtle sightings or incidents should be recorded (see Appendix A for recommended sighting form template) and reported to the *Herpetofauna Database Administrator*. It is recommended DOC's current reporting and record keeping processes be reviewed to reflect current management objectives (see section 2.7).

2.7 Recommendations

This guide is intended primarily for the emergency treatment and recovery of marine turtles for DOC field staff in the Northland region. There are a number of additional areas that would benefit from further review and development. These include but are not limited to:

Reporting and record keeping: At present, all marine turtle incidents (whether sightings or recoveries) should be reported to and maintained on the DOC *Herpetofauna Database*. However, it appears that not all reports are recorded. At the time of writing (June 2007), the author had collected 26 marine turtle records for the period: 1/01/2006 – 1/06/2007 (Appendix E), compared with 2 records held by the DOC *Herpetofauna Database* for the same period (B. Kappers, pers. comm.). Also, at present, the structure of the Herpetofauna database focuses primarily on terrestrial reptiles and therefore the data fields do not adequately reflect those required for marine turtles. It is recommended to:

- Review current reporting and record keeping procedures to ensure marine turtle data is adequately managed.
- Review the structure of the current NZ herpetofauna database to reflect appropriate data files and to also compliment other regional marine turtle databases (e.g. South Pacific Regional Environment Programme Marine Turtle Database).
- Develop a standard incident for all marine turtle encounters (e.g. Appendix A).
- Develop channels with other organisations (e.g. Ministry of Fisheries) to enable encounters to be reported directly to the appropriate administrators.
- Ensure rehabilitation facilities (e.g. Kelly Tarlton’s) report periodically (annually) on the status of marine turtles maintained at each facility. Reports should be sent to the DOC database administrator (as well as other relevant organisations) for review.

Key Contact list: It is recommended to review the current contact list where for DOC staff can obtain information for emergency veterinary advice, collaborating research organisations (national and regional), treatment and rehabilitation facilities, transport, necropsy, specimen collection and storage. Some key contacts have been added (Appendix D) but others may include: Ministry of Fisheries, Te Papa Museum, Auckland Museum, and appropriate refrigerated transport company.

Veterinarian and Field staff training: Although the emergency treatment for marine turtles is similar to other vertebrates, there are certain specific considerations and procedures that would benefit from specialised training. Also, at present, advice for the emergency treatment of distressed marine turtles is frequently sought form those listed in section 2.5. None of these contacts are currently contracted to DOC and consequently treatment and other costs incurred, are covered by those parties. Therefore it is recommended to:

- Develop a network of trained veterinarians and DOC field staff that can carry out emergency treatment in the first-instance.
- Produce an appropriate standard veterinary manual for use by DOC contracted veterinarians and rehabilitation centres. Collaboration with Kelly Tarlton’s, Lynfield Veterinary Clinic, Wildlife Health Centre – Massey University, would be advantageous.
- Train DOC staff and contracted veterinarians to ensure correct handling, treatment, and sampling procedures are maintained. Collaborating on-site with turtles held at Kelly Tarlton’s would be advantageous.
- Develop necropsy and disease investigation procedures in collaboration with Kelly Tarlton’s, Lynfield Veterinary Clinic, Wildlife Health Centre – Massey University.
- Review how treatment, transport, and other costs should be funded.
- Identify other organisations that may contribute (e.g. New Zealand Wildlife Health Centre, Massey University, Palmerston North).

Research and management: Both nationally and regionally, there are number of organisations concerned with the research, conservation, and management of marine turtles in New Zealand and the Pacific. It is recommended to identify those person(s) and institutions that are able to contribute and collaborate on the management and conservation of marine turtles in New Zealand. These links should be developed and added to the network contact list and where appropriate facilitate adequate data sharing and management. For example; the South pacific Regional Environment Programme maintains a regional database and tagging programme (SPREP flipper tag series).

Acknowledgements

This report would not have been possible without funding from the Department of Conservation, Science Advice Fund. Thanks to Tony Beauchamp for the directive to undertake this report, and Kevin Turner, Benno Kappers, and Brett Gartrell for their advice and assistance. Also, thanks to Dave Donnelly, Tony Beauchamp, and Steve Cook for reviewing drafts. Finally, thanks to Summer Lockhart for producing some of the figures.

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George, R. H. 1997: Health problems and diseases of sea turtles. Pp. 363-385 *in* Lutz, P. L; Musick, J. A. (Eds): The Biology of Sea Turtles, Volume I. CRC Press, Boca Raton, Florida. 432p.

Gill, B. J. 1997: Records of turtles and sea snakes in New Zealand, 1837-1996. *New Zealand Journal of Marine and Freshwater Research 31*:477-486.

Kemf, E. G., B.; Abreu, A.; Wilson, A. 2000: Marine Turtles in the Wild. A World Wildlife Fund Species Status Report - 2000. Gland, Switzerland. 40p.

Milton, S. L.; Lutz, P. L. 2003: Physiological and genetic responses to environmental stress. Pp. 163-197 *in* Lutz, P. L.; Musick, J. A.; Wyneken, J. (Eds): The Biology of Sea Turtles, Volume II. CRC Press. 496p.

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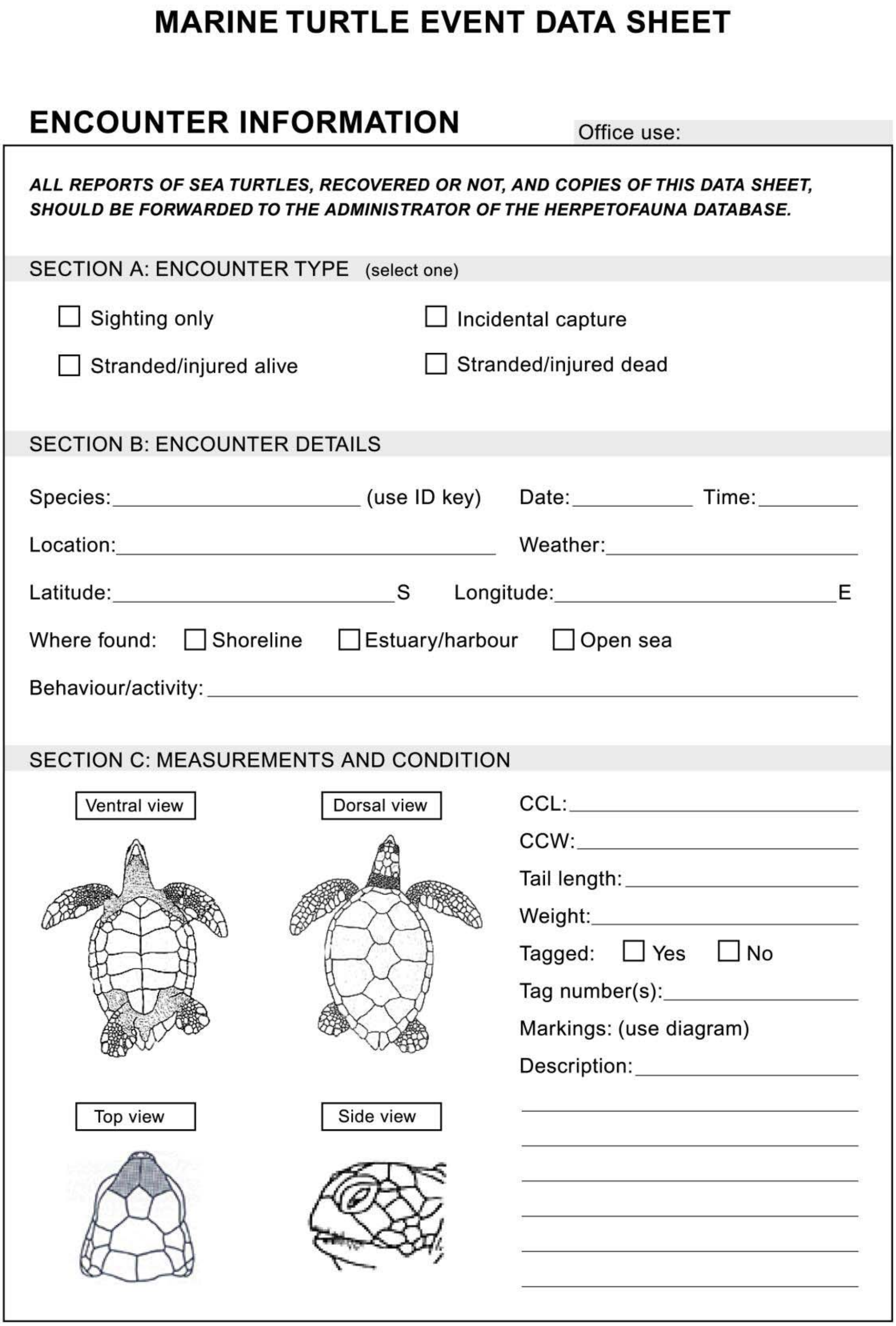
SPREP. 1989: Action Strategy for Nature Conservation in the South Pacific Region. Fourth South Pacific Conferenece on Nature Conservation and Protected Areas, September 1989, Port Vila, Vanuatu. Prepared by the South Pacific Regional Environment Programme.

Wibbels, T. 1999: Diagnosing the sex of sea turtles in foraging habitats. Pp. 139-143 *in* Eckert, K. L.; Bjorndal, K. A.; Abreu-Grobois, F. A.; Donnelley, M. (Eds). Research and Managment Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, 1999. 235p.

Wyneken, J. 2001: The Anatomy of Sea Turtles. U.S. Department of Commerce NOAA Technical Memorandum NMFS-SEFSC-470. 172p.

Appendix A: Marine turtle event data sheet



MARINE TURTLE EVENT DATA SHEET

ENCOUNTER INFORMATION

Office use:

ALL REPORTS OF SEA TURTLES, RECOVERED OR NOT, AND COPIES OF THIS DATA SHEET, SHOULD BE FORWARDED TO THE ADMINISTRATOR OF THE HERPETOFAUNA DATABASE.

SECTION A: ENCOUNTER TYPE (select one)

☐ Sighting only ☐ Incidental capture

☐ Stranded/injured alive ☐ Stranded/injured dead

SECTION B: ENCOUNTER DETAILS

Species: ____________ (use ID key) Date: ________ Time: ________

Location: ____________________ Weather: ____________

Latitude: ____________ S Longitude: ____________ E

Where found: ☐ Shoreline ☐ Estuary/harbour ☐ Open sea

Behaviour/activity: ______________________________

SECTION C: MEASUREMENTS AND CONDITION

CCL: ____________

CCW: ____________

Tail length: ____________

Weight: ____________

Tagged: ☐ Yes ☐ No

Tag number(s): ____________

Markings: (use diagram)

Description: ____________

SECTION D: ACTIONS TAKEN

Action taken:____

Turtle: ☐ Alive ☐ Dead Turtle recovered: ☐ Yes ☐ No

Assessed for hypothermia: ☐ Yes ☐ No Class rating: ☐ 1 ☐ 2 ☐ 3 ☐ 4

Sent to:____

Reported to:____

Images taken: ☐ Yes ☐ No Sent to:____________________

Samples taken: ☐ Yes ☐ No DNA sample taken: ☐ Yes ☐ No

Sent to:____

IF ENTANGLED: ***PHOTOGRAPH THE ENTANGLED ANIMAL PRIOR TO UNTANGLING AND KEEP A SAMPLE OF THE DEBRIS FOR FURTHER IDENTIFICATION.***

SECTION E: INCIDENTAL CAPTURE (Commercial only)

Name of vessel: ____________________ Registered MSA number:____________

Target species: __________________ Fishing method: __________________

Description of capture:____________________________________

__________________________________ Captured: ☐ Alive ☐ Dead

SECTION F: INCIDENTAL CAPTURE (Non-commercial only)

Fishing method:__________________________ Captured: ☐ Alive ☐ Dead

Description of capture:____________________________________

____

SECTION G: OBSERVER DETAILS

Name:____

Organisation: ____

Address: ____

Phone:__________________ Email: ______________________________

NOTES

____

____

____

____

____

Appendix B: Marine turtle field kit

Marine turtle event data sheet (Appendix A).

Note book (or waterproof paper) and pencil.

Waterproof marker pen.

First aid kit including:

- Clean gauze or cloth (for covering open wounds).
- Saline solution (for cleaning wounds).
- Large syringe – no needle (for spraying saline).
- 1:10 dilution of povidone-iodine solution.
- Antibiotic ointment.

Tissue sampling kit (Appendix C).

Stretcher, tarpaulin, or mat.

Thermometer (suitable for cloacal readings; see Appendix F).

Water based, water-soluble lubricant (e.g. *K-Y Jelly: Johnson & Johnson*).

Flexible tape measure.

Camera.

Latex examination gloves.

Protective clothing.

Knife and scissors.

Sample bags and bottles.

Labels.

Clean towel.

Flipper tags, tag applicator.

Appendix C: Collecting tissue samples

Where tissues samples are required for genetic analysis, the method described below can be used by field staff to collect samples from live and dead marine turtles.

Biopsy kit includes:

Record sheet, waterproof labels (2), and pencil. Sample vials (2) with 70% ethanol preservative. Scalpel blade (2), latex gloves, alcohol (2) and iodine (2) swabs, tweezers, parafilm, and plastic bag.

Tissue source:

Live animals: skin

Dead animals: skin or muscle

Method:

- Put on latex gloves before handling animal. For live animals, turn onto shell (back) to immobilise. **Do not handle turtle by flippers – use secure grip of shell edge (front and back) to handle.**
- Swab biopsy area (live animals: loose skin around neck or flippers) with alcohol.
- Collect 2 samples of approximately 1 cm^2 skin (for live animals, only uppermost layer of skin should be taken to avoid bleeding) or muscle tissue biopsy using tweezers and scalpel blade. Note: if more than one animal is to be sampled, use a new sterile blade and only one sample tube per animal to avoid cross contamination.
- Place samples into separate tubes, write date on both sides of labels with pencil, and place in tubes with samples.
- If live animal, swab area with iodine.
- Wrap parafilm around vial caps (to prevent leaking), place samples in plastic bag, seal, and send together with record sheet to address above.

Note:

- Samples can be kept at ambient temperature for at least one year, but avoid exposure to heat or sunlight. Samples are best kept in fridge until shipment, if possible.
- If live animals are caught offshore, and are not injured or compromised, return to sea after samples are taken. If animal is injured or compromised, notify 0800 DOCHOT line, and hand over to DoC office as soon as possible for rehabilitation.
- For all animals found stranded or incidentally caught coastally, DoC should be notified immediately for assessment.

Appendix D: Recommended contact list

Emergency Treatment and Recovery Advice

Kevin Turner - veterinarian
Lynfield Veterinary Clinic
122 White Swan Rd
Mt. Roskill, Auckland
Ph. 09 626 4335
Mobile: 027 727 96560
Email: lynvet@paradise.net.nz

Dr Brett Gartrell
New Zealand Wildlife Health Centre
Massey University
Palmerston North
Ph. 06 356 9099 ext 7398
Email: B.Gartrell@massey.ac.nz

Dan Godoy
Earth & Oceanic Science Research Institute
Auckland University of Technology
Private Bag 92006, Auckland 1020
Mobile: 021 742 656
Email: dgodoy@aut.ac.nz

Rehabilitation and Treatment Facilities

Kelly Tarlton’s Antarctic Encounter and Underwater World
Andrew Christie – Aquarium Curator
23 Tamaki Drive
Orakei, Auckland
Phone: 09 528 0603 or DDI: 09 528 1720

Mobile: 021 222 1266

Email: andrew.christie@kellytarltons.co.nz

Website: www.kellytarltons.co.nz

National Aquarium of New Zealand

Rob Yarrall – Operations Manager

Marine Parade, Napier

Phone: 06 834 1404

Fax: 06 833 7631

Email: rob@nationalaquarium.co.nz

Website: www.nationalaquarium.co.nz

South Pacific Regional Environment Programme

Lui Bell

Marine Species Officer

Secretariat of the Pacific Regional Environment Programme

PO Box 240

Apia, SAMOA

Ph. +685 66281 or +685+ 21929 Ext 281

Mobile: +685 777-2939

Fax: +685 20231

Email: luib@sprep.org

Website: www.sprep.org

Anne Trevor

Turtle Database Officer

Secretariat of the Pacific Regional Environment Programme

P.O. Box 240

Apia, SAMOA

Ph. +685 21929

Fax: +685 20231

Email: annet@sprep.org

Website: www.sprep.org

Appendix E: New Zealand marine turtle records (1/1/2006 – 1/6/2007)

No.	Encounter code	Common name, *Species*	Date found/ sighted	Sighting type	Location	Region
1	NZCM0306/010	Green *Chemyd*	05-Mar-06	Sighting only	Poor Knights Is.	Northland
2	NZUN0306/011	Unconfirmed	12-Mar-06	Sighting only	Rangaunu Hbr	Northland
3	NZDC0306/012	Leatherback *Dercor*	18-Mar-06	Sighting only	Bream Bay, Whangarei	Northland
4	NZCM1106/019	Green *Chemyd*	04-Nov-06	Sighting only	Deep Water Cove, B.O.I.	Northland
5	NZCC0107/023	Loggerhead *Carcar*	12-Jan-07	Sighting only	Stephens Island, Marlborough Sounds	Nelson/ Marlborough
6	NZDC0107/024	Leatherback *Dercor*	27-Jan-07	Sighting only	Walker Rock, Marlborough Sounds	Nelson/ Marlborough
7	NZUN0207/025	Green *Chemyd*	10-Feb-07	Sighting only	Rangiputa	Northland
8	NZCM0507/028	Green *Chemyd*	19-May-07	Sighting only	Rangiputa	Northland
9	NZDC0206/009	Leatherback *Dercor*	04-Feb-06	Incidental capture - Recreational	Whangaparoa Peninisula, Auckland	Auckland
10	NZCM0406/013	Green *Chemyd*	01-Apr-06	Incidental capture - Recreational	Meola Reef - Pt. Chevalier, Waitemata Harbour	Auckland
11	NZCM0307/026	Green *Chemyd*	07-Mar-07	Incidental capture - Recreational	Whatuwhiwhi	Northland
12	NZCM0106/008	Green *Chemyd*	29-Jan-06	Incidental capture - Commercial	Unknown format (decimal?): -34.8 S 173.3	NE NZ
13	NZDC0906/017	Leatherback *Dercor*	07-Oct-06	Incidental capture - Commercial	Unknown format (decimal?): -29.2 S 179.9	NE NZ
14	NZDC1106/020	Leatherback *Dercor*	09-Nov-07	Incidental capture - Commercial	Unknown format (decimal?): -31.2 S 180.6	NE NZ
15	NZDC1106/021	Leatherback Dercor	09-Nov-07	Incidental capture - Commercial only	Unknown format (decimal?): -31.2 S 180.7	NE NZ

No.	Encounter code	Common name, *Species*	Date found/ sighted	Sighting type	Location	Region
16	NZDC0106/007	Leatherback Dercor	12-Jan-06	Stranded/ injured - Dead only	De Lamberts Bch, Golden Bay	Nelson/ Marlborough
17	NZCM0506/14	Green *Chemyd*	21-May-06	Stranded/ injured - Dead	Henderson Bay	Northland
18	NZCM0407/027	Green *Chemyd*	03-Apr-07	Stranded/ injured - Dead	Stanley Bay	Auckland
19	NZCM--/033	Green *Chemyd*	??	Stranded/ injured - Dead	Waitemata Hbr	Auckland
20	NZCM0706/015	Green *Chemyd*	22-Jul-06	Stranded/ injured - Recovered alive	Rangiputa Bch	Northland
21	NZCM0806/016	Green *Chemyd*	21-Aug-06	Stranded/ injured - Recovered alive	Rangiputa Bch	Northland
22	NZCM1006/018	Green *Chemyd*	11-Oct-06	Stranded/ injured - Recovered alive	90 Mile Bch	Northland
23	NZCM1206/032	Green *Chemyd*	03-Dec-06	Stranded/ injured - Recovered alive	Mahuta Point, West Coast	Northland
24	NZCM0107/022	Green *Chemyd*	05-Jan-07	Stranded/ injured - Recovered alive	Ocean bch, Tairua	Coromandel
25	NZCM0507/029	Green *Chemyd*	26-May-07	Stranded/ injured - Recovered alive	Whangarei Hbr	Northland
26	NZCM0507/030	Green *Chemyd*	30-May-07	Stranded/ injured - Recovered alive	Whangarei Hbr	Northland

Appendix 3

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
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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 were 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 were 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, Figure 1). Only 12 (*n* = 10%) records occurred in regions off the South Island, with all from FMA 7 (Figure 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, Figure 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 (Figure 1). This is particularly relevant to leatherback turtles where 98% (*n* = 88) of all bycatch records of this species occurred in oceanic habitats (Figure 2). In contrast, green turtles were most often captured in neritic habitats i.e. over the continental shelf in water depths < 200 m (Figure 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 (Figure 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 (Figure 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 (Figure 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).

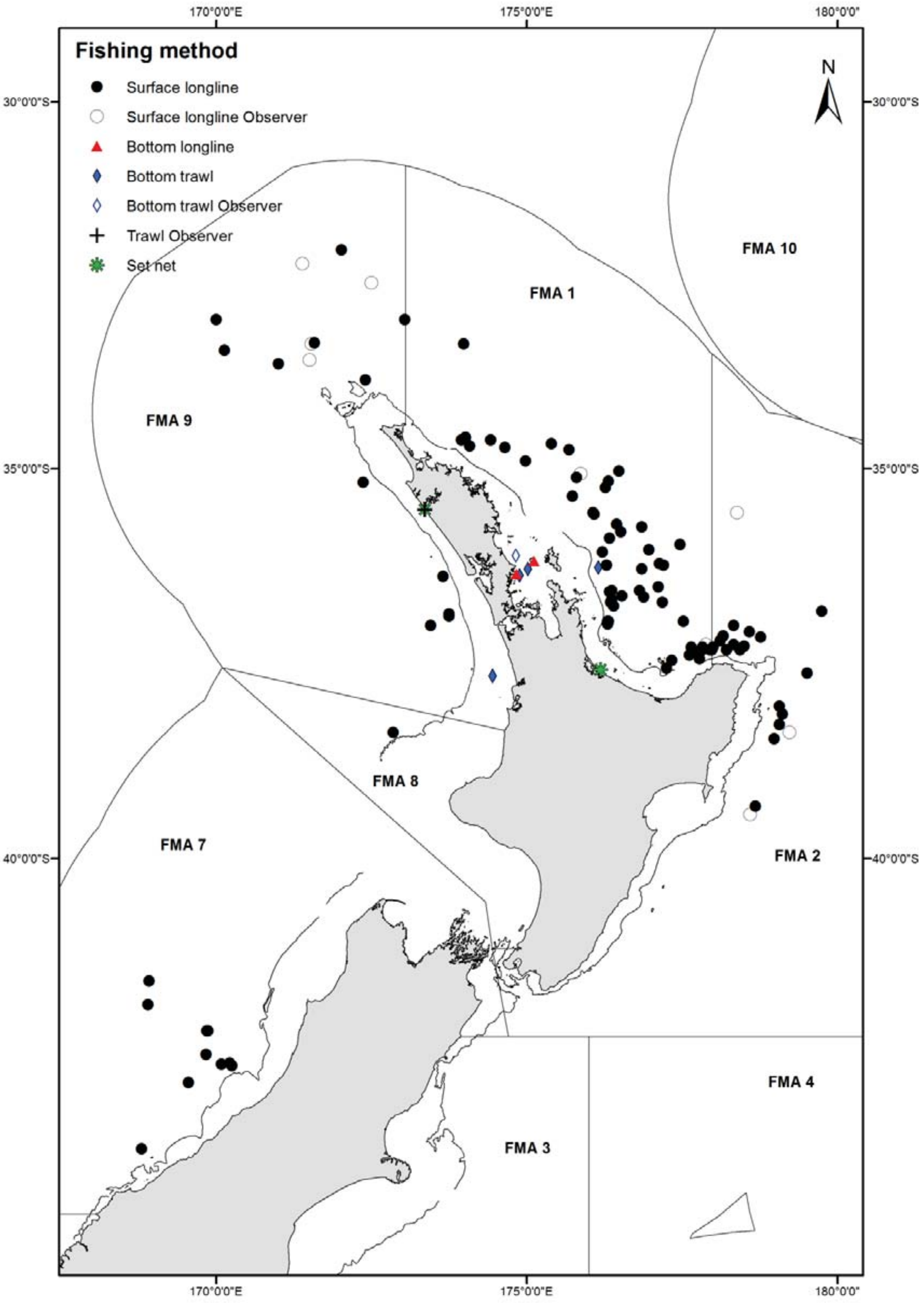

Figure 1. Distribution of all reported marine turtle bycatch data from 1 July 2008 to 30 November 2015 (n = 120). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area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

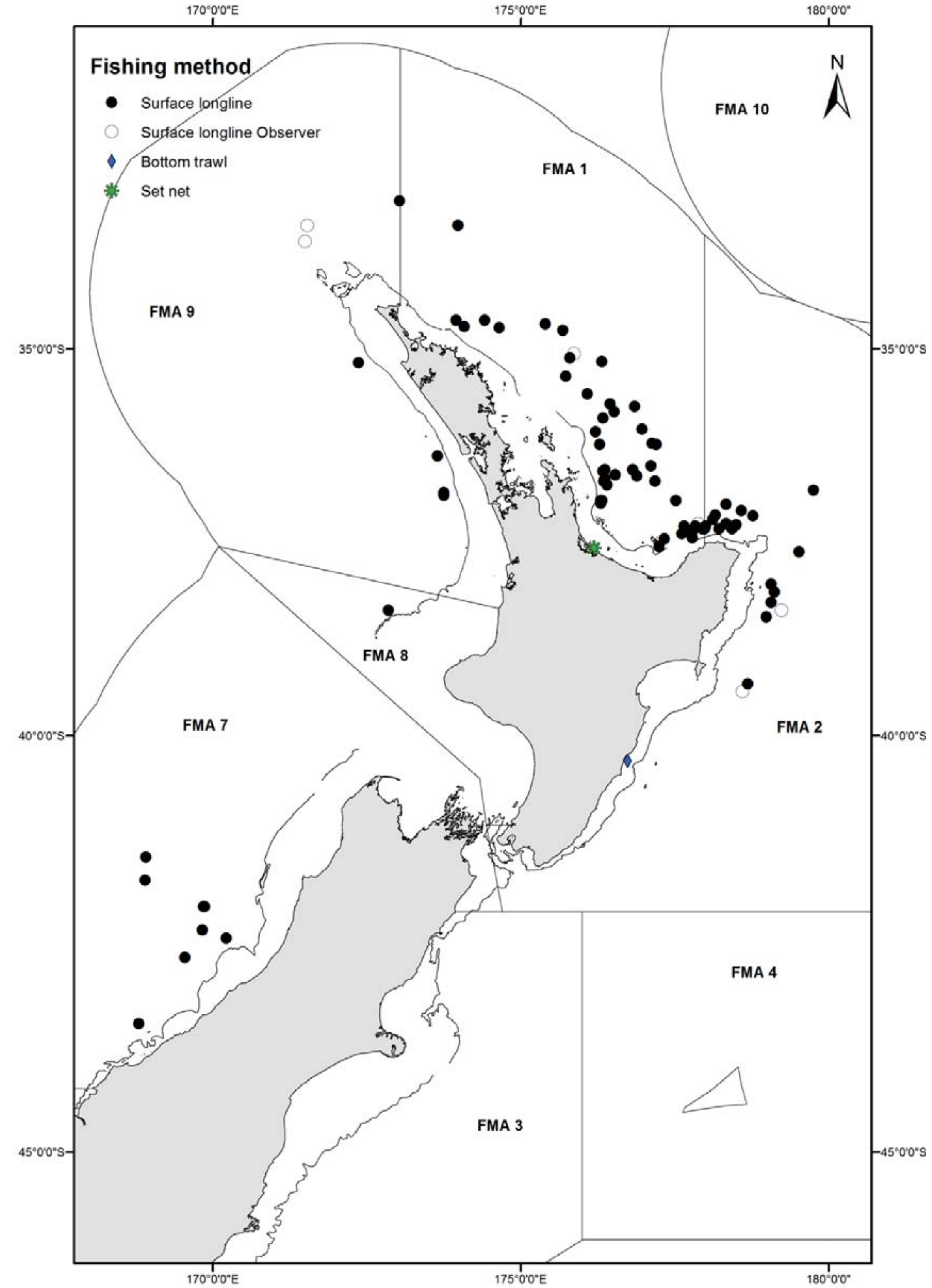

Figure 2. Distribution of reported leatherback turtle bycatch from 1 July 2008 to 30 November 2015 (n = 90). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA)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

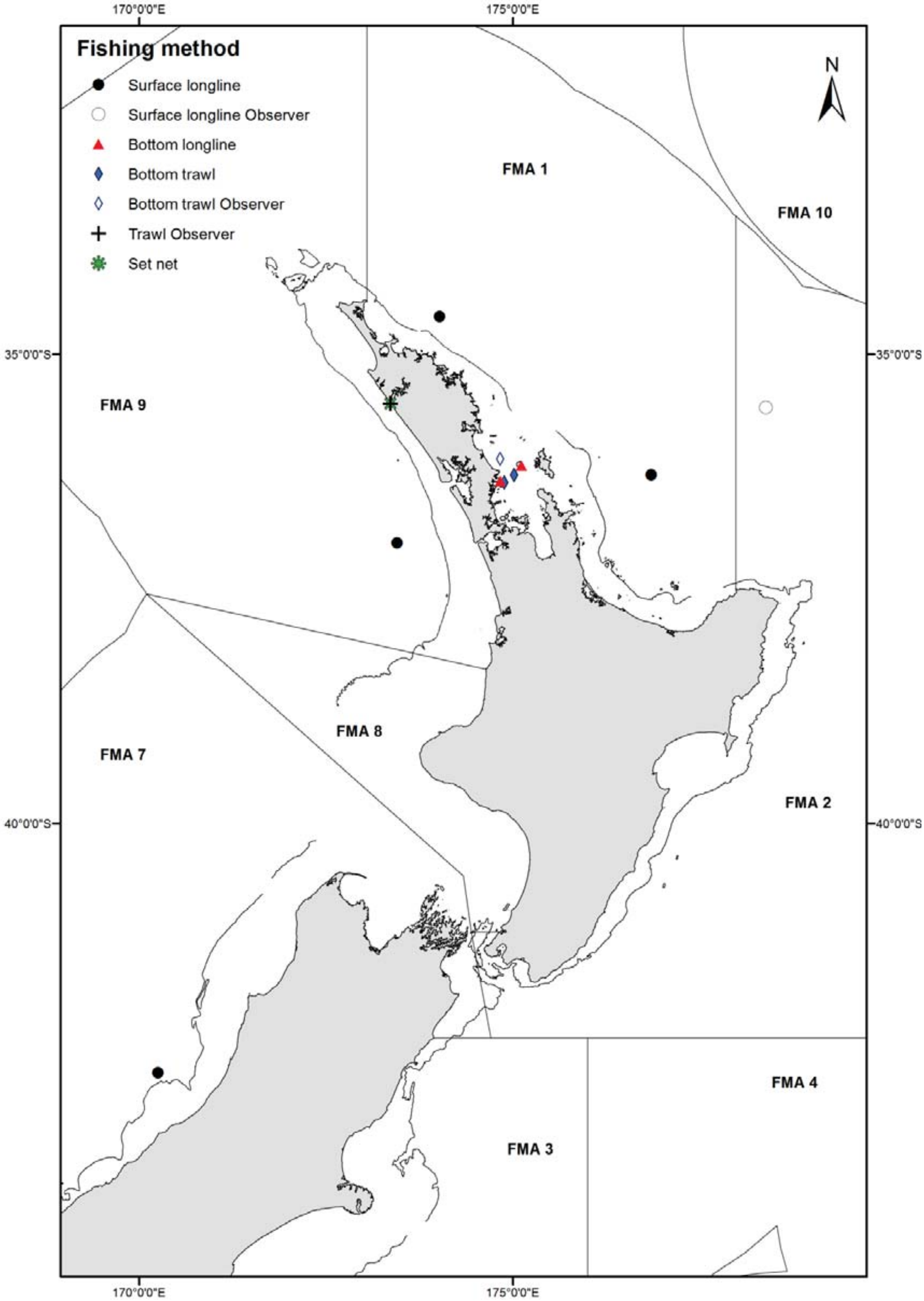

Figure 3. Distribution of reported green turtle bycatch from 1 July 2008 to 30 November 2015 (n = 12). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries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

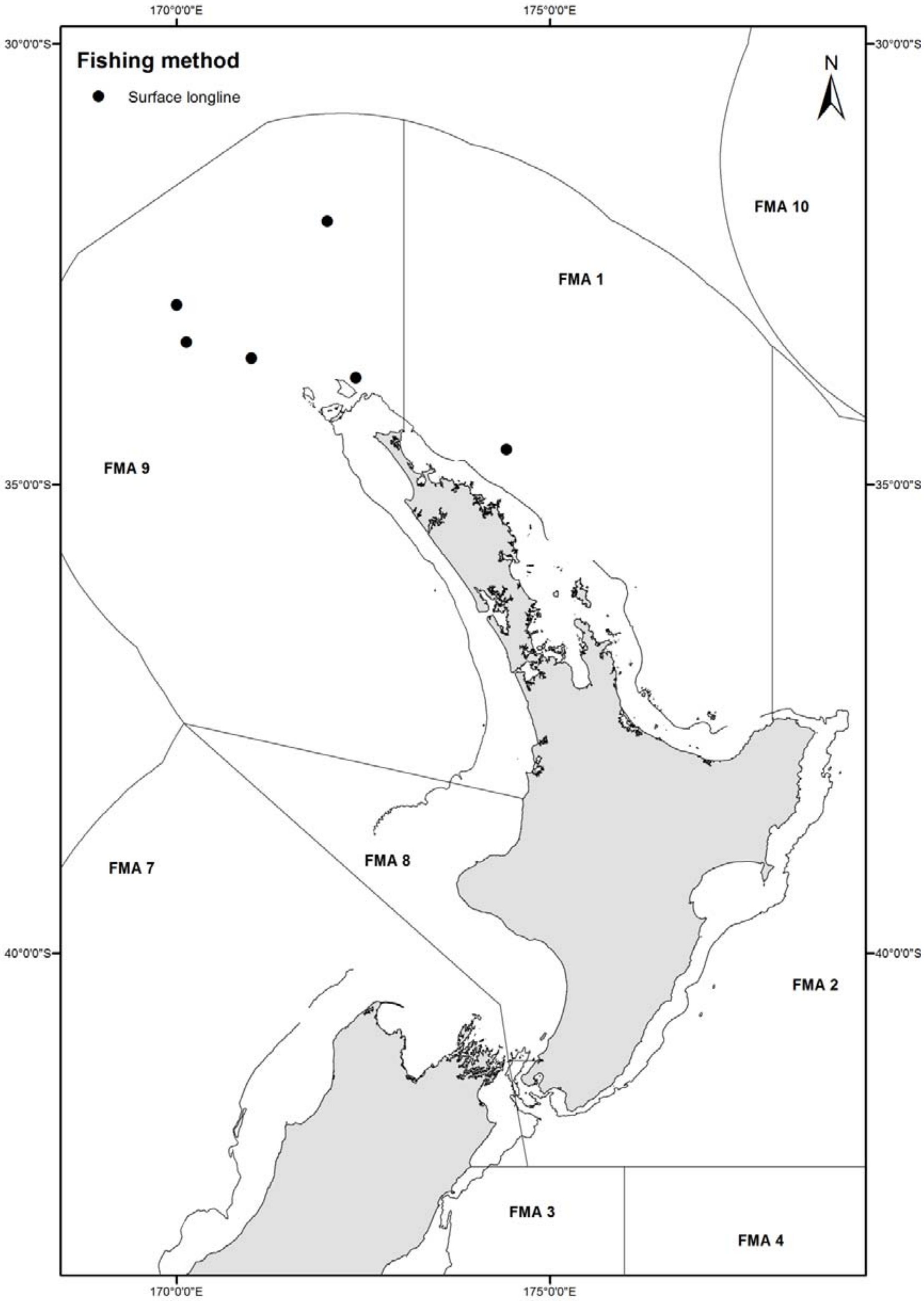

Figure 4. Distribution of reported hawksbill turtle bycatch from 1 July 2008 to 30 November 2015 (n = 6). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries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

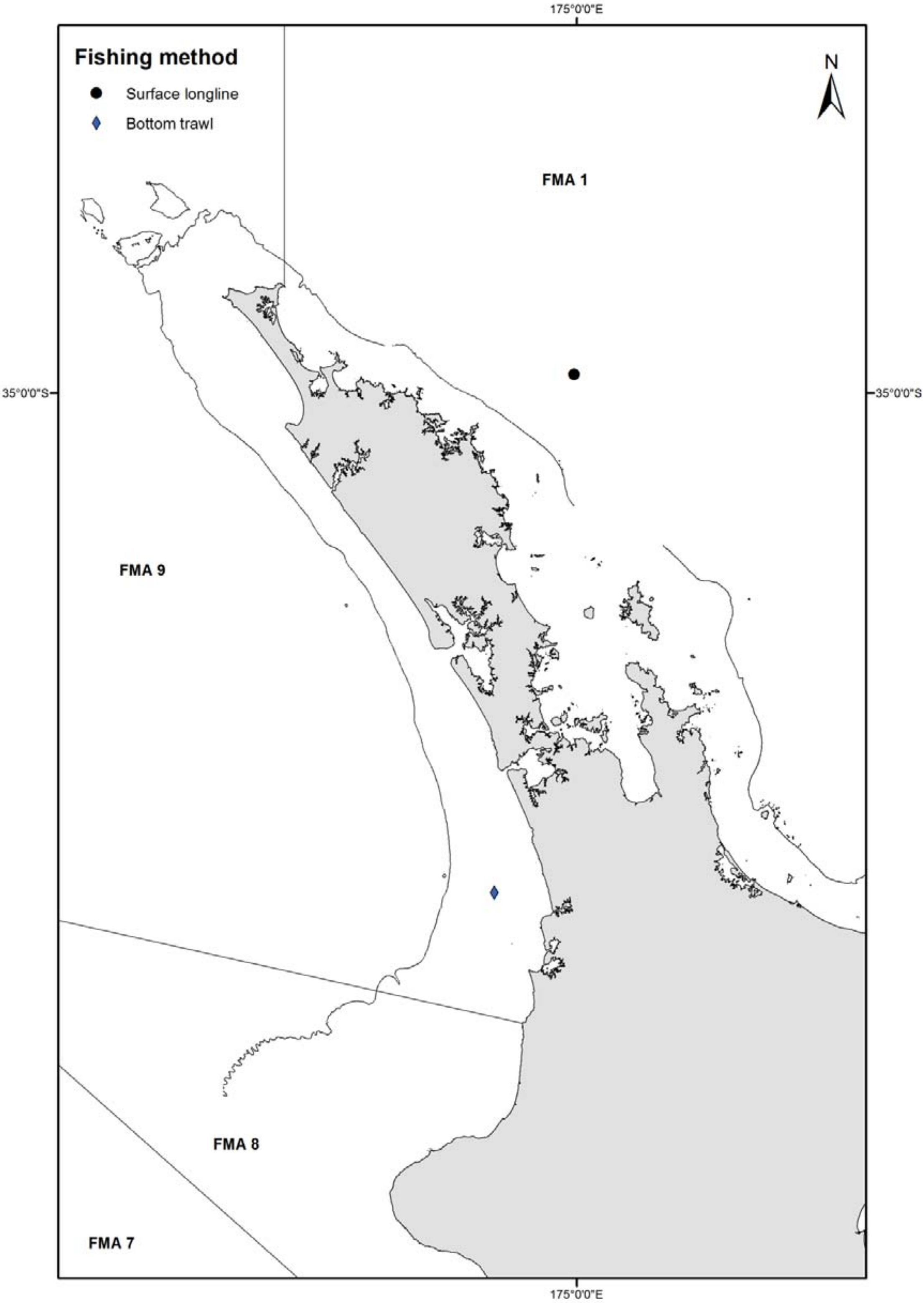

Figure 5. Distribution of reported loggerhead turtle bycatch from 1 July 2008 to 30 November 2015 (n = 2). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) boundaries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

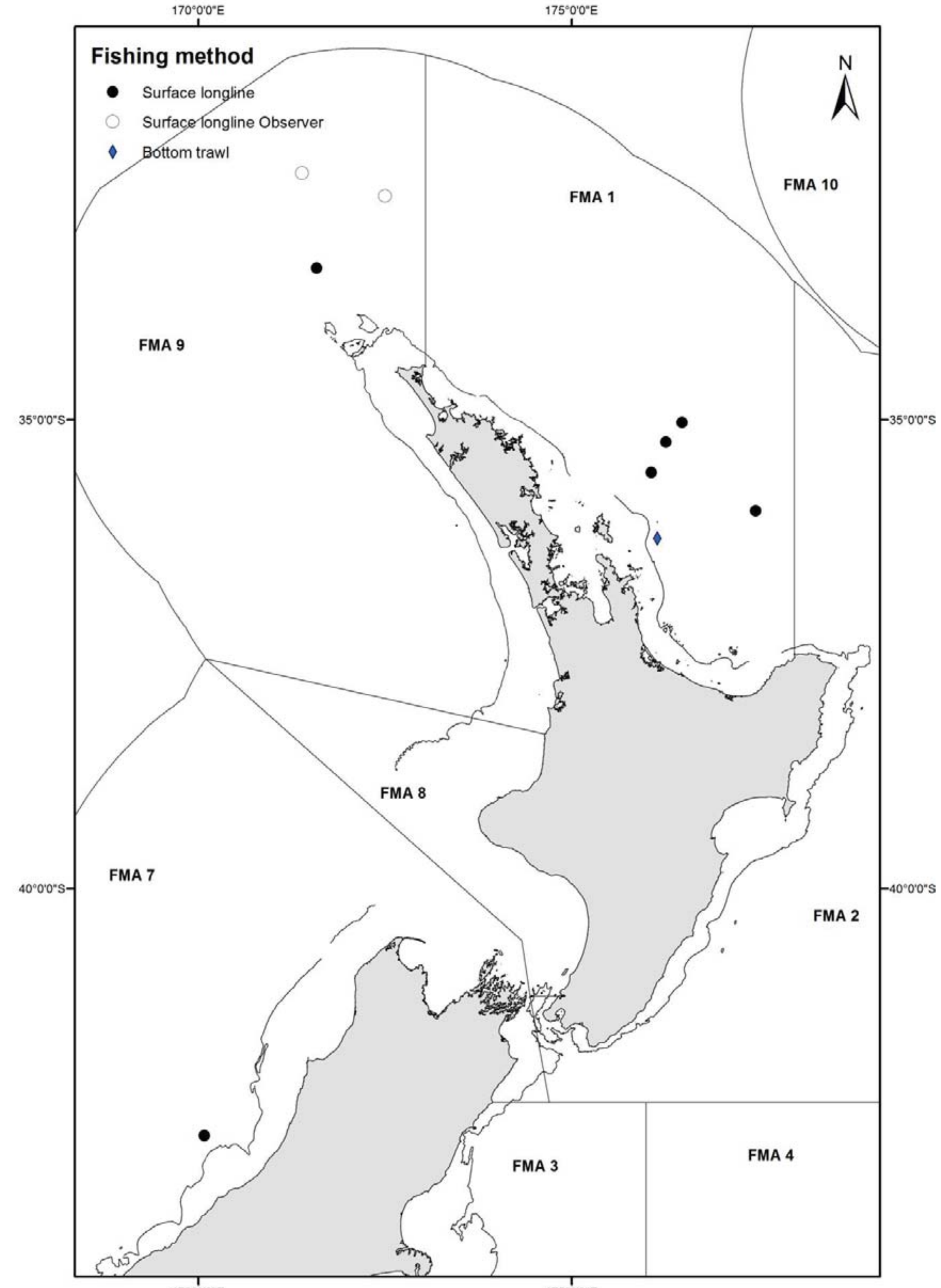

Figure 6. Distribution of reported unidentified turtle bycatch from 1 July 2008 to 30 November 2015 (n = 10). Data includes commercial and observer records across all reported fishing methods. The 200 m continental isobath, New Zealand Exclusive Economic Zone (EEZ), and Fisheries Management Area (FMA) 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; Table 7) 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 (Table 7). 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.

Table 7. 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).

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 (ca. 71° N) and as far south as New Zealand (Foveaux Strait: ca. 47° 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 ° 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 (Kaplan 2005; Donoso and Dutton 2010; Wallace et al. 2011; B P Wallace et al. 2013; Curtis et al. 2015).

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 (ca. 30^{o} S) south to Foveaux Strait (ca. 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 ca. 230 kg (Pritchard & Mortimer 1999). Overall generation length for this species have been estimated at ca. 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 (ca. 30^{o} S) south to Canterbury (ca. 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 (ca. 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 ca. 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 (ca. 30^{o} S) south to Palliser Bay, Wellington (ca. 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 ca. 180 kg (Pritchard & Mortimer 1999; Dodd 1988). Overall generation length for this species has been estimated at ca. 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 ca. 25 cm CCL in the Mediterranean, 46-64 cm in the western Atlantic, ca. 60 cm in Japan, and ca. 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 (ca. 30^o S) south to Stewart Island (ca. 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 ca. 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 (ca. 35^o S) south to Stewart Island (ca. 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.

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).

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).

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.

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.

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.

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