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FALSE KILLER WHALES
(*PSEUDORCA CRASSIDENS*)
IN NEW ZEALAND WATERS



A thesis presented in partial fulfilment of the requirements for the degree of

Master of Science

in

Conservation Biology

at Massey University, Albany, New Zealand

Jochen Reinhard Zaeschmar

2014

Abstract

On a global scale, false killer whales (*Pseudorca crassidens*) remain one of the lesser-known delphinids, with most knowledge about the species originating from infrequent stranding reports. Herein I examine the occurrence, site fidelity and association patterns of false killer whales in the waters of northeastern New Zealand from historic stranding records (1870 to date) and at-seas observations collected between 1995 and 2012.

Despite the infrequency of strandings (28 recorded events in 144 years), false killer whales are among the most numerous cetaceans to strand on New Zealand shores due to large mass strandings events. Stranding records suggest that the species occurs predominantly in North Island waters, with only 7% of strandings ($n = 2$) recorded on South Island shores. Hawkes Bay and the Chatham Islands appear to be stranding hotspots for the species in New Zealand.

At-sea encounters support occurrence in northern waters, with all sightings reported in waters off northeastern New Zealand. False killer whales were infrequently encountered in the study area; however, of the 61 distinctive photo-identified individuals, 88.5 % ($n = 54$) were resighted, with 70.5 % ($n = 43$) resighted on three or more occasions, with two individuals observed eight times. Eighty-five percent ($n = 52$) were observed in more than one year and at least two individuals were resighted almost seven years after their initial identification, with movements as far as 650 km documented for eight individuals.

Results indicate that all false killer whales photo-identified in the study area so far, are linked in a single social network. Group sizes ranged from 20 to *ca.* 150 ($\bar{x} = 46.7$, $SD = 28.48$). Distance from shore and bottom depth ranged from < 1 to 67.4 km ($\bar{x} = 9.16$, $SD = 14.85$) and 25 to 350 m ($\bar{x} = 105.33$, $SD = 86.66$) respectively, with most records in shallow (< 100 m) continental shelf waters. Occurrence in these nearshore waters is likely seasonal, with all sightings between December and May, coinciding with the shoreward flooding of a warm current during that period. Abundance estimates indicate that the number of individuals that occurred in the study area during the sampling period may be as low as 111. While some individuals exhibited injuries consistent with detrimental fishery interactions, these were scarce ($n = 2$), with no new injuries recorded since 2007.

Additionally, interspecific associations between false killer whales and common bottlenose dolphins (*Tursiops truncatus*) are described, with the two species observed together during 91.5 % ($n = 43$) of encounters. These mixed-species groups occurred across the entire range of the study area and during all encounter months. Group sizes for common bottlenose

dolphins within these mixed-species groups ranged from 5 to *ca.* 250 ($\bar{x} = 62.8$, $SD = 42.79$). Photo-identification shows repeat inter- and intraspecific associations among individuals. Thirty-four percent ($n = 51$) of common bottlenose dolphins photo-identified in mixed-species groups with false killer whales had repeat associations with false killer whales, with 28.2 % ($n = 42$) resighted in such groups in more than one year. Individuals were observed together with false killer whales up to 1832 days (*ca.* 5 years, $n = 2$) after association was initially recorded and across a range as far as 650 km ($n = 1$). While foraging was observed during 39.5% ($n = 17$) of mixed-species encounters, observed predation by killer whales (*Orcinus orca*) and close interspecific interactions suggest that anti-predatory and social factors may also play a role in the formation of these mixed-species groups.

This study represents the first long-term record of seasonal resightings of this species in New Zealand waters and suggests the likelihood of a small local population. A precautionary approach in the assignment of the species' appropriate conservation status is therefore recommended.

Acknowledgements

I would like to sincerely thank my supervisor Dr. Karen Stockin for taking me on and believing in the false killer whale project. I appreciate the opportunity and all the support you have given me. Thank you also to my co-supervisor Dr. Emmanuelle Martinez for tirelessly and promptly answering any question and to Dr. Ingrid Visser for getting the ball rolling.

This project is the result of a great collaborative effort of commercial tour operators and fellow researchers. A sincere thank you goes out to my fellow Massey University researchers Sarah Dwyer, Anna Meissner, Catherine Peters and Tania Gaborit-Haverkort and the Orca Research Trust's Dr. Ingrid Visser for sharing their data with me. This thesis would not be the same without your contributions. The study could not have been realised without the support from whale- and dolphin-watch operators from around New Zealand. Thank you to everyone who contributed sighting reports, photographs and a wealth of information on false killer whales in their areas.

In the Bay of Islands; Fullers Great Sights for allowing access to their historic whale-watch records; the crew of the *Tutunui*, in particular Joanne 'Floppy' Halliday and Tammy Johnston; Rob Hunt and Gemma Langsdale from Explore Images, Sue Hews from Magic Memories, William Goodfellow and Kit Nixon from Explore New Zealand and Vanessa McKay from Carino Dolphin Sail. On the Tutukaka coast; Kate Malcolm from Dive Tutukaka. In the Hauraki Gulf; the crew of Dolphin Explorer, in particular Andy Light. In the Bay of Plenty; Stuart Arnold, Rosie Roess and Cameron Fines and Helen Cadwallader from Dolphin Seafaris. Graeme Butler from *Gemini Galaxsea*, Patrick O'Sullivan from PeeJay White Islands tours. In Kaikoura; Dennis Buurman from Dolphin Encounter, Jody Fish and Alastair Judkins from the Kaikoura Ocean Research Institute (K.O.R.I.). In Christchurch; Cally Burke and Chris Collins from Heritage Expeditions.

The following operators kindly participated in the presence/absence of sightings survey: Dolphin Rendezvous, Carino Dolphin Sail, Dolphin Discoveries, Great Sights, Dive Tutukaka, Dolphin Explorer, Butler's Gemini Galaxy, Dolphin Seafaris, Pee Jay White Island Tours, Dolphin Watch Ecotours, Golden Future, Dolphin Encounter, Whale Watch Kaikoura, World of Whales, Black Cat Cruises, Real Journeys, Monarch Wildlife Cruises, Fiordland Expeditions, Aihe Eco Charters, Rakiura Charters, Kaikoura Helicopters, Pelorus Tours, Heritage Expeditions. Thank you for sharing your information with me.

The following people supplied additional photographs: John Allan, Annika Andresen, Rebecca Bellini, Alex Black, Robert Bradley, Tom Brough, Sonja Clemens, Matthew Coutts, Mary Cowling, David Donnelly, Clinton Duffy, Mandy Everett, Sarah Gardner, Margaret Ginnelly, Cathy Goeldner, Daniel Goetz, Monica Greco, David Hall, Tommy Hatwell, Lara Kay, Lily Kozmian-Ledward, Karin Neumann, Daniel Panek, Taira Peters, Mazdak Radjainia, Richard Robinson, Jean Paul Rouen, Nathan Turner and Elisa Weiss. Thank you!

A special mention goes to Dr. Robin Baird. Thank you for all your support and good advice and for being a fountain of knowledge on all things *Pseudorca*.

Thank you to Phil Crawford from Greenpeace for background information and use of their false killer whale image, to Sakae Hemmi and the Elsa Nature Conservancy for use of their image and to Dr. Simon Ho, Dr. Tom Gilbert and Dr. Robin Baird for use of figures.

Thank you to Dr. Gabriela de Tezanos-Pinto, Dr. Emmanuelle Martinez and Jens Currie for help with statistical analysis.

Thank you to the Department of Conservation; Elke Reufels for permit assistance and operator liaisons and Hannah Hendriks for compiling records of the New Zealand Stranding database.

Thank you to Aquatic Mammals and Marine Mammal Science for permission to include their research articles in this thesis.

Thank you to my ‘fin-fatigued’ friend Jo Berghan for tireless fin matching, meticulous reviewing, good chats and yummy meals. Sarah Dwyer and Catherine Helen Peters for help with just about anything, and Lily Kozmian-Ledward for mutual support and cheering up. My fellow whale rescuers Joanne ‘Floppy’ Halliday and Dr. Ingrid Visser for all sorts of personal and logistical support. Millie Coleing for help with reviewing. Tom Brough, Marta Guerra and Anna Meissner for plenty of good advice and James Ross for financial and practical support.

Lastly, immense thanks go to my dad, Gerhard Zaeschmar for his continued support and to my husband, Geir Sveaass for endless patience, encouragement and understanding.

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Chapter One

General introduction



A juvenile and an adult false killer whale off northeastern New Zealand

1.1 Introduction

The false killer whale (*Pseudorca crassidens*) is a top marine predator (Perryman and Foster 1980, Palacios and Mate 1996). The role of top (or apex) predators in structuring ecosystems has long been acknowledged (Estes *et al.* 2001) and they are often considered keystone species due to the disproportionally large impacts they may have on communities or ecosystems (Barua 2011). Despite some historical disagreement (*e.g.* Caughley 1970) it is now widely accepted that a decline in such predators is associated with a wider loss of biodiversity (Estes *et al.* 2001).

Top predators are thought to regulate biodiversity both directly via controlling the abundance and composition of the prey community (top-down effects, Estes *et al.* 2001) and indirectly, through effecting anti-predator behaviour in its prey (*risk effect*, Heithaus *et al.* 2008). For example, the nocturnal upward migration of mesopelagic fauna into productive surface waters and the subsequent return to deeper waters during daylight hours is driven by the avoidance of sight-dependent surface predators (Lampert 1989). As such top predators may not only influence ecosystem composition but also affect population traits on evolutionary timescales (Estes *et al.* 2001).

Top predators may also initiate top-down processes and trophic cascades. These processes have been well documented in terrestrial habitats. For example, studies of North American ecosystems show that the extirpation of top predators such as gray wolves (*Canis lupus*) and grizzly bears (*Ursus arctos*) could be attributed to a marked increase in ungulate numbers, which in turn led to a marked decline in woody plant communities (Beschta and Ripple 2009). The resulting trophic cascade was shown to have a broad scale impact on the respective ecosystems, affecting a wide range of other taxa such as birds, insects and amphibians, and even initiated landscaping processes through increased stream erosion (*e.g.* Berger *et al.* 2001, Beschta and Ripple 2009).

In marine ecosystems, the effects of the extirpation of top predators are also well known. For example, exploitation of upper-trophic level sharks in the Northwest Atlantic caused a cascading top-down effect, leading to an eight-fold increase in the number of cownose rays (*Rhinoptera bonasus*), which in turn effected a crash in the local bay scallop (*Argopecten irradians*) population (Baum and Worm 2009).

The ecosystem services provided by cetaceans can be much harder to detect and little is known about their potential impact on prey resources, likely owing to the logistical

difficulties of their study due to their mobility, sparseness and cryptic nature. The killer whale (*Orcinus orca*) may be a possible exception, with the most famous case possibly being the trophic cascade that led to the decline of Alaskan kelp forests (Estes *et al.* 1998, Williams *et al.* 2004). Over-exploitation of seals and sea lions in Alaska is attributed to a shift in killer whale diet to sea otters (*Enhydra lutris*), resulting in a drastic decline of the sea otter population. In turn, this decline caused an increase in sea urchin (*Strongylocentrotus* spp.) numbers, the sea otters' main prey, which resulted in the decimation of local kelp forests and the subsequent degradation of the ecosystem. This example shows the complexities of predator-prey interactions and further highlights that the role that top predators play in ecosystems may be highly context dependent. Consequently, it remains debated whether ecosystem functions are essentially controlled by a few key species (the keystone species model, *e.g.* Paine 1966) or by a wider and more complex process of foodweb interactions (Polis and Strong 1996).

While it remains uncertain to what extent respective predators affect biodiversity, it is clear that they are often linked to high biodiversity, not solely because of the ecosystem services they provide but also due to the biodiversity they require (Sergio *et al.* 2008). As such they are considered suitable sentinel species (or condition indicators) for the identification of conservation needs, given that their high position in the food web together with their life history traits (*e.g.* slow reproduction, low density, specialised diet etc.) make them particularly vulnerable to ecosystem changes. It follows that apex predators are often used to identify sites in need of protection (Sergio *et al.* 2008). Indeed, top predators are frequently used as indicator species both in terrestrial (*e.g.* Sergio *et al.* 2008) and aquatic systems (*e.g.* whales, sharks and seabirds, Boyd and Murray 2001).

The demanding habitat requirements, both spatially and ecologically, of most top predators has led to their increasing utilization as umbrella species (Simberloff 1998), given that their appropriate protection simultaneously protects large areas and a wide range of associated flora and fauna. A wide variety of examples exist, both from terrestrial and aquatic systems (*e.g.* Barua 2011), including cetaceans (Hooker and Gerber 2004). However, these factors can also be the very reasons that may hinder effective protection, as the areas and processes that would be affected often present considerable logistical challenges and may result in significant economic impacts. Top predators may also serve indirect conservation purposes and conservation managers have long since recognised that the charismatic nature of many top predators is a highly effective means to raise both public awareness and funds for conservation issues (Sergio *et al.* 2008). Many terrestrial and aquatic examples, illustrating the use of top predators as so called flagship species, exist in the literature (*e.g.* Sergio *et al.*

2006, Clucas *et al.* 2008). Perhaps the main benefit of using top predators as umbrella species may be the fact that their substantial spatial requirements may inspire wildlife managers to “think large” when designing protected areas (Ray *et al.* 2005), thereby protecting a wider range of species and habitats (Rosenzweig 1995). The study of the ecology and demographics of an apex predator such as the false killer whale helps to elucidate the species’ ecosystem services and requirements, which in turn may facilitate not only appropriate management, but also wider conservation benefits.

The false killer whale remains one of the lesser-known members of the delphinid family, despite a global distribution in tropical and warm temperate waters (Stacey *et al.* 1994), a circumstance likely owing to its predominant pelagic distribution (Baird 2008). With the exception of a dedicated research programme in Hawaiian waters (*e.g.* Baird *et al.* 2008), current knowledge of the species is primarily derived from captive specimens as well as from incidental sightings and strandings (*e.g.* Leatherwood *et al.* 1989). Likewise, information about the species from New Zealand waters remains scarce and primarily anecdotal (*e.g.*, Gaskin 1972, Cawthorn 1986, Cawthorn 1991), with most of the peer reviewed publications to date (Visser *et al.* 2010, Zaeschmar *et al.* 2013, Zaeschmar *et al.* 2014, Appendices B, C and D), resulting from the research reported here. The present study provides a first assessment of false killer whale occurrence in New Zealand waters, incorporating aspects of seasonality, site fidelity and interspecific associations in order to further our knowledge and understanding of this little known species. It is envisaged that this thesis will provide the necessary data to assist managers in determining adequate management strategies and conservation measures. This introductory chapter provides a literature review, summarizing current scientific knowledge of false killer whales, with particular emphasis placed on the respective aspects addressed in this thesis.

1.2 Taxonomy

The false killer whale is one of the larger members of the delphinid subfamily Globicephalinae. It was first described from a sub-fossil skeleton, recovered from the Lincolnshire fens in England by Owen (1846) who placed the species in the genus *Phocaena*, (with *crassidens* meaning thick-toothed). The species was subsequently classified as *Orca* (Gray 1846). Presumed to be an extinct species, the common name was derived from the similarities in skull morphology to that of the killer whale (Baird 2008). The other common name ‘blackfish’ is not exclusive to the false killer whale but was historically used to describe various members of the subfamily Globicephalinae such as pilot whales (*Globicephala* spp.). With the first live specimen documented in 1861 (Mobius

1862) and with other specimens becoming available around that time (Reinhard 1862), it became evident that the false killer whale was an extant species that was neither *Phocaena* nor *Orcinus*. Consequently, Reinhard (1862) placed the species in its own genus *Pseudorca*. *P. crassidens* is the only member of its genus and there are no subspecies recognised.

LeDuc *et al.* (1999) reported that the false killer whale is most closely related to the Risso's dolphin (*Grampus griseus*), the pygmy killer whale (*Feresa attenuata*), the melon-headed whale (*Peponocephala electra*), pilot whales (*Globicephala* spp.) and the killer whale. More recently, McGowen *et al.* (2009) and Vilstrup *et al.* (2011) reconfirmed LeDuc *et al.*'s findings, with the exception of placing the killer whale as a more distant relative (Figure 1.1). Substantial differences in the cranial characteristics between false killer whales from Australia, South Africa and Scotland led to the proposal and subsequent abandonment of two distinct southern and northern hemisphere forms (Hector 1873).

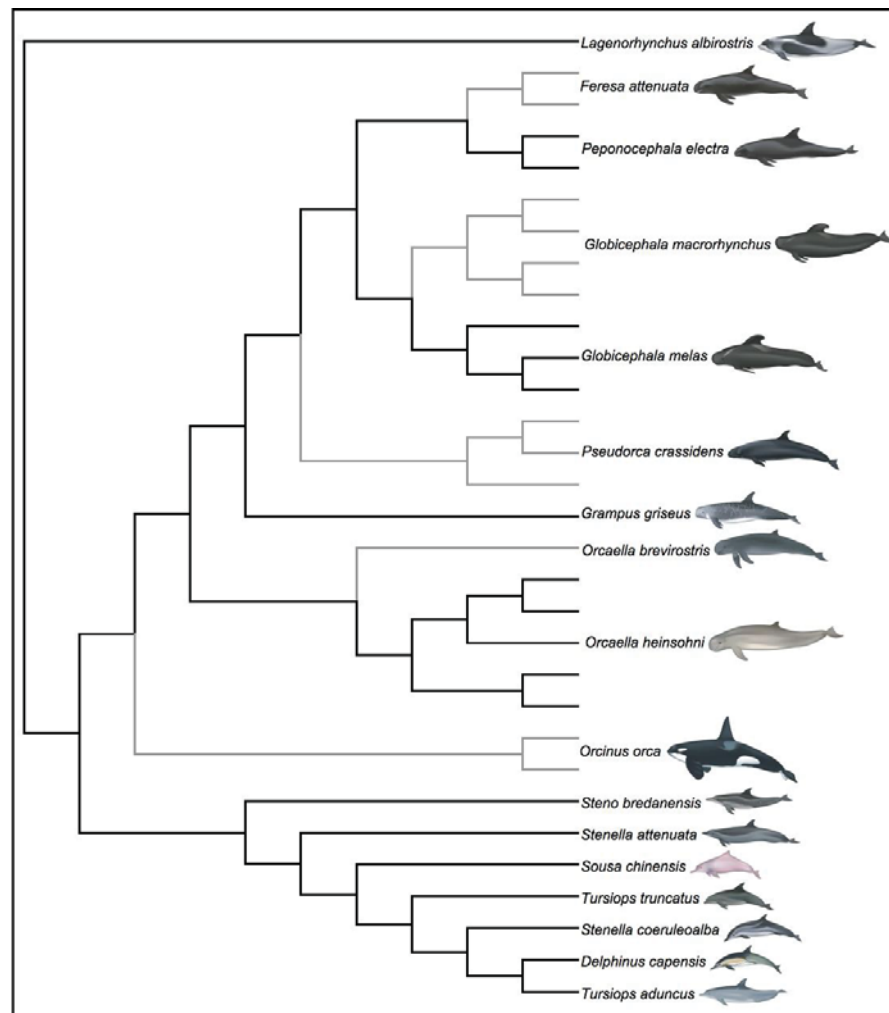


Figure 1.1. Phylogenetic tree of selected taxa within Delphinidae, showing the false killer whale's close relatedness to other globicephalids, in particular the long-finned pilot whale and Risso's dolphin. Figure based on Vilstrup *et al.* 2011. Courtesy of Simon Ho.

1.3 Morphology



Figure 1.2. The false killer whale showing its characteristic slender body, small rounded head and slightly pointed falcate dorsal fin. The lighter ‘cape’ posterior of the head is only visible in some individuals. Source: Reeves *et al.* (2002).

The false killer whale is slender bodied with a relatively small rounded head and no beak (Odell and McClune 1999, Figure 1.2). Almost uniformly black or dark grey, the most noticeable colour variation is a light grey to almost white anchor shaped patch on the ventral surface that starts at the throat and can proceed as far as the genital slit (Norris and Prescott 1961, Kitchener *et al.* 1990). Additionally, some animals display an area of grey on both sides of the head demarcated by a dark bridle (Perrin 1997). The slightly pointed falcate dorsal fin, located near the mid-point of the body, ranges from *ca.* 20 to 40 centimeters (cm) in height (Leatherwood *et al.* 1989, Stacey *et al.* 1994). The pectoral fin may measure up to 55 cm in length and is characterised by a distinct ‘hump’ on the leading edge (Leatherwood *et al.* 1989). The teeth are large and conical, ranging from 7 to 11 in each upper jaw and from 8 to 12 in the lower jaws (Purves and Pilleri 1978).

Despite a continuous global distribution, some notable morphological differences between false killer whales from different regions have been reported. Mean body size can differ significantly in adult individuals from different areas (Ferreira *et al.* 2014), and variations in skull and external morphology between individuals from Australia, Scotland and South Africa have been documented (Kitchener *et al.* 1990, Ferreira *et al.* 2014). Furthermore, genetically closed populations identified in Hawaiian waters (Chivers *et al.* 2007, Martien *et al.* 2014) show that localised, disjunct false killer whale populations exist.

1.4 Range and distribution

1.4.1 Global range and distribution

False killer whales inhabit tropical to warm temperate seas worldwide and occasionally venture into cold temperate waters (Baird *et al.* 1989, Figure 1.3). The species is thought to occur in a continuous range between 50° N and 50° S (Odell and McClune 1999), with the majority of sightings reported from lower latitudes, generally between 40° N and 40° S. Surveys from the eastern North Pacific show a marked decrease in density north of 15° N, further supporting a preference for warmer waters (Ferguson and Barlow 2003). Likewise, the reported water temperature range for the distribution of false killer whales lies between 9 and 30.8° C, although the species is thought to be more common in waters towards the higher end of that range (Stacey *et al.* 1994). A comprehensive overview of false killer whale sightings is provided in Table 1.1.

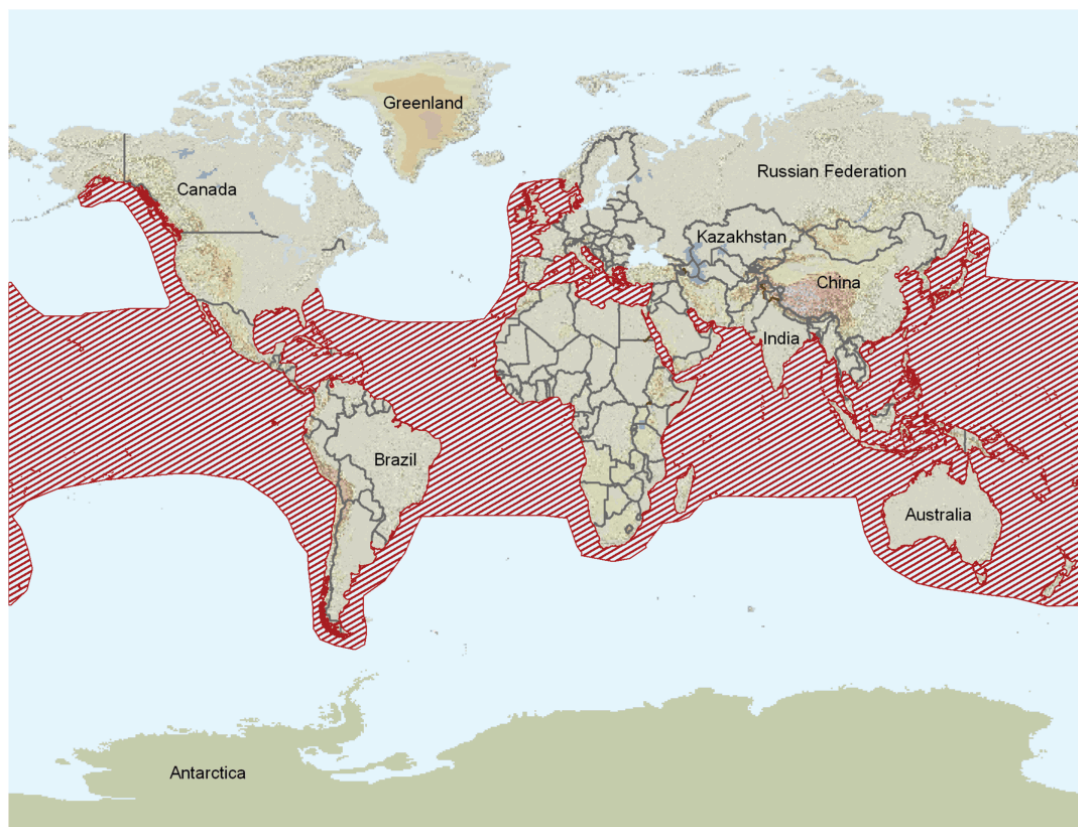


Figure 1.3. World map showing the assumed global range of false killer whales (red lined area).
Source: IUCN

Table 1.1 Annotated list of false killer whale sightings, documenting the species' range in the three main ocean basins and their associated seas. (Note: St/As refer to strandings and at-sea observations).

Location	Nearest land mass	St/As	Source
<u>Atlantic Ocean</u>			
Western North Atlantic			
	Cape Hatteras, North Carolina	St	Brimley 1937
Gulf of Mexico	Texas	As	Caldwell and Caldwell 1973
	Florida	St	Odell <i>et al.</i> 1979
	Louisiana	As	Mullin and Fulling 2004
	Louisiana to Florida	As	Maze-Foley and Mullin 2006
	Offshore Gulf of Mexico	As	Jefferson & Schiro 1997
	Northern Gulf of Mexico	As	Vazquez Castan <i>et al.</i> 2009
Caribbean	St. Vincent	As	Caldwell <i>et al.</i> 1971
	Venezuela	St/As	Romero <i>et al.</i> 2001
	Puerto Rico	As	Mignucci-Giannoni 1998
	Colombia	St/As	Fraija <i>et al.</i> 2009
	Aruba	As	Luksenburg 2011
	Cuba	As	Whitt <i>et al.</i> 2012
Eastern North Atlantic			
	Cape Verde	As	Hazevoet <i>et al.</i> 2010
	Azores	As	Silva <i>et al.</i> 2003, Gomes Pereira 2008
	Canaries	St	Hernandez Garcia 2002
	Morocco	As	Nortarbartolo di Sciara 1997
	Ghana	As	Van Waerebeek 2009
Bay of Biscay	France	As	Brereton <i>et al.</i> 2001
North Sea	Scotland	St	Matheson and Cowley 1934
	England	St	Evans 1980
	Denmark	St	Reinhard 1866
Irish Sea	Wales	St	Evans 1980
Baltic Sea	Denmark	St	Reinhard 1866
Mediterranean			
	Catalonia, Spain	As	Pilleri 1967
	Balearic Islands	As	Duguy and Cyrus 1973
	Cyprus	As	Boisseau <i>et al.</i> 2010
	Syria	St	Kasperek 1997
	Egypt	St	Wassif 1956
	Israel	As	Kerem <i>et al.</i> 2012
Strait of Gibraltar	Spain	As	Casinos and Vericad 1976
	Ligueran Sea	As	Duguy and Cyrus 1973
Adriatic Sea	Italy		Bearzi <i>et al.</i> 2004
Aegean Sea	Greece	As	Frantzis <i>et al.</i> 2003
Eastern South Atlantic			
	Angola	As	Weir 2011, Weir <i>et al.</i> 2013
	Gabon	As	Van Waerebeek and De Smet 1996,
	Cote d'Ivoire		Weir <i>et al.</i> 2013
			Weir <i>et al.</i> 2013
	Namibia	As	Findley <i>et al.</i> 1992
	South Africa	As+St	Findley <i>et al.</i> 1992, Best 2007, Best and Reeb 2010, Kirkman <i>et al.</i> 2010
Western South Atlantic			
	Brazil	St	Andrade <i>et al.</i> 2001
	Argentina	St	Langguth 1977
Strait of Magellan	Chile	St	Alonso <i>et al.</i> 1999
<u>Pacific Ocean</u>			
Eastern North Pacific			
	Alaska	St	Leatherwood <i>et al.</i> 1988
	Mexico	As	Meraz and Sanchez-Diaz 2008
	Costa Rica	As	Acevedo Guitierrez <i>et al.</i> 1993, May-Collado 2005, Martinez-Fernandez <i>et al.</i> 2010

Table 1.1 cont'd

Location	Nearest land mass	St/As	Source
	Colombia	As	Herrera-Carmona <i>et al.</i> 2011
	Hawaii	As	Baird <i>et al.</i> 2012
	Palmyra Atoll	As	Barlow <i>et al.</i> 2008
Western North Pacific			
	Japan	As	Miyashita 1993, Kishiro and Kasuya 1993
Sea of Japan	Japan	As	Mizue and Yoshida 1961, Kasuya 1985, Kasuya 1986
East China Sea	Japan	As	Kishiro and Kasuya 1993
	China	As	Zhou <i>et al.</i> 1995, Tseng <i>et al.</i> 2011
	Taiwan	As	Chou 1994, Chou <i>et al.</i> 2004
Yellow Sea	China	As	Zhou <i>et al.</i> 1995, Zhou <i>et al.</i> 2008
Bohai Sea	China	As	Zhou <i>et al.</i> 1995
Yangtze River	China	As	Zhou <i>et al.</i> 1995
Guan He River	China	As	Zhou <i>et al.</i> 1995
South China Sea	Brunei	As	De Boer 2000
	Cambodia	As	Beasley and Davidson 2007
	Vietnam	As	Smith <i>et al.</i> 1995
	Thailand		Chantrapornsyil <i>et al.</i> 1996
	Malaysia		Ponnampalem 2012
Eastern South Pacific			
	Marquesas	As	Gannier 2002, Gannier 2008
	Society Islands	As	Gannier 2008
	Galapagos Islands	As	Palacios and Mate 1996
	Ecuador	As	Castro 2004, Alava <i>et al.</i> 2005
	Chile	As	Aguayo <i>et al.</i> 1998, Flores <i>et al.</i> 2003
Patagonian Fjords	Chile	As	Viddi <i>et al.</i> 2010
Western South Pacific			
	Palau	As	Miyazaki and Wada 1978
	Guam	As	Fulling <i>et al.</i> 2010
	Salomon Islands	As	Takekawa 2002
	New Caledonia	As	Greaves 1998, Garrigue and Greaves 2001
	Samoa	As	Johnston <i>et al.</i> 2008
	New Zealand	As+St	Baker 1981, Brabyn 1991, Cawthorn 1979, Cawthorn 1981, Cawthorn 1986, Cawthorn 1991
Tasman Sea	Australia	St	Phillips 1988
	New Zealand	St	Baker 1981, Brabyn 1991
Bass Strait	Tasmania	St	Scott and Green 1975
Eastern North Indian Ocean			
	India	As	James and Mohan 1987
	Indonesia	As	Kreb and Budiono 2005, Borsa and Nugroho 2010
Timor Sea	Indonesia	As	Barnes 1991, Rudolph <i>et al.</i> 1997
	Australia	As	Palmer <i>et al.</i> 2009
Western North Indian Ocean			
	Maldives	As	Anderson 2005
	La Reunion	As	Poisson and Taquet 2001
	Seychelles	As	Bargain <i>et al.</i> 2002
	Somalia	As	Ballance and Pitman 1998
Persian Gulf		St	Braulik 2010
Red Sea		As	Baldwin <i>et al.</i> 1998
Gulf of Aqaba		As	Beadon 1991 in Baldwin <i>et al.</i> 1999
Eastern South Indian Ocean			
	South Africa	As	Findlay <i>et al.</i> 1992
	Mozambique	As	Kiszka <i>et al.</i> 2007
Western South Indian Ocean			
	Southwest Australia	St	Chambers and James 2005
Great Australian Bight	South Australia	St	Aitken 1971

Despite one of the largest continuous ranges among cetacea (Davies 1963), false killer whales are encountered infrequently in most locales where they occur (Leatherwood *et al.* 1989, Odell and McClune 1999). Table 1.2 provides examples of available abundance estimates from various regions, further highlighting the species' scarce distribution. As a predominantly pelagic species that was never hunted commercially on a notable scale (Gaskin 1968), most of our past understanding of false killer whale distribution has been derived from stranding records. Unfortunately, historically, reliable at-sea sightings have been relatively scarce and sometimes hindered by confusion and/or deliberate pooling with other 'blackfish' (globicephalids, *e.g.* Kasuya 1971, Brabyn 1991).

Table 1.2. Examples of false killer whale density and abundance estimates from North Pacific regions. (Note all locations for which data could be obtained relate to the North Pacific.)

Area	Size	Individual density per 100 km ²	Estimated population	CV (%)	Source
Western North Pacific	2,220,707 km ²	0.07	16,668	0.26	Miyashita 1993
Oceanic northern Gulf of Mexico	380,432 km ²	0.27	1,038	0.71	Mullin and Fulling 2004
Exclusive Economic Zone surrounding Hawaii	2,240,024 km ²	0.06	1,503	0.66	Bradford <i>et al.</i> 2012
Eastern tropical Pacific	191,480,000 km ²	0.03	39,800	0.64	Wade and Gerrodette 1993

However, in recent years there has been a noticeable influx in at-sea sighting reports from all three large ocean basins, confirming a wide, yet relatively scarce distribution. Furthermore, frequent interactions with commercial fishing operations in offshore waters (*e.g.*, Nishida and Shiba 2002, Ramos-Cartelle and Mejuto 2008) have also improved our understanding of false killer whale occurrence in remote regions. Whilst considered primarily pelagic in habit (Odell and McClune 1999), false killer whales are known to venture close to shore at oceanic islands (Acevedo-Gutiérrez *et al.* 1997, Garrigue and Greaves 2001, Gannier 2002, Baird *et al.* 2008) and may advance far into continental shelf waters (Weir *et al.* 2013) where they have been observed to enter enclosed harbours (Palmer *et al.* 2009) and even rivers (Zhou *et al.* 1995).

Baird *et al.* (2008) suggested that false killer whales in Hawaiian waters use the greatest range of water depths of all cetaceans surveyed in their study, a behaviour possibly based on the wide ranging movements and diverse habitats of their prey. Satellite tagged individuals of the Hawaiian island-associated population did not venture further than 122 kilometres (km) from shore (median = 11.7 km) throughout the duration of the study, indicating that

strong site fidelity and limited home ranges exist at least in some populations (Baird *et al.* 2012).

Despite a few observations at high latitudes such as Prince William Sound, Alaska (Leatherwood *et al.* 1988), Dornoch Firth, Scotland (Matheson & Cowley 1934), or the Strait of Magellan (Alonso *et al.* 1999), most of those records relate to stranded individuals and may therefore be non-representative of the species normal range (Leatherwood *et al.* 1989). Long-distance travel (Tomilin 1957) and changes in seasonal occurrence have been observed or suggested in various locations (Kasuya 1985, Kasuya 1971, Findlay *et al.* 1992, Acevedo-Gutiérrez *et al.* 1997, Palmer *et al.* 2009), and movements of false killer whales have been linked to warm water masses and migrations of prey (Tomilin 1957, Kasuya 1971). However, it is not clear if, or to what extent, populations of the species migrate.

1.4.2 Occurrence in New Zealand waters

Little is known about false killer whale distribution in New Zealand waters, with the majority of records in the region relating only to strandings (Baker 1981, Brabyn 1991). The first such recorded stranding event, being of a single individual, occurred near Wellington in 1870 (Hector 1873, Gaskin 1972). Since, strandings have been reported as far south as *ca.* 45°S in the Pacific and *ca.* 37°S in the Tasman Sea, albeit infrequently (Brabyn 1991). False killer whales are rarely observed at sea (Gaskin 1972, Baker 1983), with only a handful of mostly anecdotal sighting reports in existence (Cawthorn, 1979, Cawthorn, 1981, Cawthorn, 1986, Cawthorn, 1991, Gaskin 1972), highlighting the need for dedicated research on this species in the region.

1.5 Life history

Female false killer whales reach sexual maturity between the ages of 8 and 11 (Purves and Pilleri 1978, Ferreira *et al.* 2014), possibly up to 10 years prior to their male counterparts (Kasuya 1986, Ferreira *et al.* 2014). Purves and Pilleri (1978) estimated the gestation period at 15.5 months, while Ferreira *et al.* (2014) calculated a range from 14.1 to 15.7 months, very similar to that of pilot and killer whales (Purves and Pilleri 1978, Marino 1997). A seasonal peak in breeding has been suggested by Kasuya (1986) who calculated a mating peak around March to April for false killer whales in Japanese waters. While the evidence for this is considered weak (Ferreira *et al.* 2014), the discovery of seasonal peaks of progesterone levels in two captive individuals indicate that reproductivity may have an inherent seasonal aspect (Atkinson *et al.* 1999).

Pregnancy rates calculated from stranded and/or beach driven individuals in Scotland, South Africa and Japan suggest that false killer whales may have a much lower reproductive rate than other Globicephalids (Purves and Pilleri 1978, Ferreira *et al.* 2014). Pregnancy rates in females ranged from 11.4 to 15% (Purves and Pilleri 1978, Kasuya 1986, Ferreira *et al.* 2014) compared to a pregnancy rate of 32% for short-finned pilot whales (*Globicephala macrorhynchus*) (Kasuya and Marsh 1984) and 37-40% for long finned pilot whales (Purves and Pilleri 1978, Martin and Rothery 1993). Kasuya (1986) calculated the calving interval at 6.9 years.

Females give birth to a single young measuring between 155 and 175 cm (Ferreira *et al.* 2014), while maximum recorded length has been documented at 6.10 and 5.06 metres (m) for males and females, respectively (Leatherwood and Reeves 1983, Perrin and Reilly 1984). Data on maximum body weight are rare but Leatherwood *et al.* (1989) note that the maximum adult weight is at least 1,360 kg, while Reidenberg and Laitman (2008) state that it may be as much as 2,500 kg.

Longevity has been estimated at 57.5-58.5 years for males and 62.5-63.5 years for females (Ferreira *et al.* 2014). While the life expectancy for females is almost identical to that of female pilot- (63 years, Kasuya and Marsh 1984) and killer whales (62 years, Olesiuk *et al.* 1990), males tend to live comparatively longer (29 and 45 years for male pilot- and killer whales, respectively). The reason for this remains unknown. As with pilot- and killer whales, a considerable post-reproductive life-span has been suggested (Marsh and Kasuya 1986, Olesiuk *et al.* 1990, Ferreira 2008). Studies of stranded females from South Africa and beach-driven females from Japan showed a very low likelihood of females being reproductive after the age of 45, suggesting a potential post-reproductive life span of more than 15 years (Ferreira 2008).

Sexual dimorphism is evident (Baird 2008), with the melon of mature males protruding farther forward than that of females (Figure 1.4). Tomilin (1957) further notes that the dorsal fin is positioned more anterior in males than in females. Carrier *et al.* (2002) suggest that the comparatively larger melon in males of some odontocete species, including the false killer whale, originates in its use during displays of male-male aggression. While male-male aggression has been inferred for other globicephalids (*e.g.* short-finned pilot whales, Oremland *et al.* 2010), such behaviour has not been documented in false killer whales to date.



Figure 1.4. An adult male false killer whale characterised by the protruding melon. Bay of Islands 2005. Photo Jochen Zaeschmar.

1.6 Behavioural ecology

As with other oceanic globicephalids, few detailed accounts of false killer whale behaviour exist in the literature, likely owing to their predominant oceanic distribution and the ensuing logistical challenges associated with their study. Most of our knowledge on the species to date is derived from captive specimens (*e.g.* Brown *et al.* 1966, Pryor 1973, Clark and Odell 1999, Delfour and Marten 2001), focusing in particular on the species' acoustic (*e.g.* Au *et al.* 2005, Nachtigall and Supin 2008, Kloepper *et al.* 2012) and swimming (*e.g.* Fish 1998, Rohr *et al.* 2002) behaviour.

Data on free-ranging individuals remain scarce and mostly pertain to *ad libitum* observations (*e.g.* Pilleri 1967, Perryman and Foster 1980, Palacios and Mate 1996), with the island-associated Hawaiian population being a notable exception (*e.g.* Baird *et al.* 2008, Baird *et al.* 2010). However, there has been a slow influx of observational false killer whale data from other regions in more recent years (*e.g.* Palmer *et al.* 2009, Weir *et al.* 2013).

1.6.1 Social structure

1.6.1.1 Group size and composition

On a global scale, false killer whale social structure remains poorly documented, with most of our detailed knowledge gained to date, derived from the dedicated studies in Hawaiian

waters (*e.g.* Baird *et al.* 2008). Described as gregarious, false killer whales typically occur in groups of 20 to 100 animals (Baird 2008), with groups as large as 600 or more individuals reported (Leatherwood *et al.* 1989). While average group sizes may vary between regions and/or populations (Ferreira 2008), significant differences in group sizes have also been reported between stranding events and at-sea observations within the same region (*e.g.*, Best 2007, Ferreira 2008, Kirkman *et al.* 2010). However, it remains unknown whether groups involved were of the same or different populations. Examples of group size differences between regions, but also between strandings and at-sea observations within regions, are illustrated in Table 1.3.

Table 1.3. Examples of differences in false killer whale group sizes from various studies of stranding events (St) and at-sea (As) observations and in different regions listed according to latitude.

Mean	Median	Range	Latitude	Type	Location	Source
18	n/a	1-89	n/a	As	Eastern Trop. Pacific	Odell and McClune 1999
35	23.5	10-100	5°N-3°S	As	Gulf of Guinea	Weir <i>et al.</i> 2013
19	17	n/a	4°N-8°S	As	Angola	Weir 2011
16	n/a	5-34	5-8°N	As	Costa Rica	Acevedo Gutierrez <i>et al.</i> 1997
125	n/a	100-150	13°S	As	Mozambique	Kiszka <i>et al.</i> 2007
15.6	16	1-35	20°N	As	Hawaii	Baird <i>et al.</i> 2013
77	55	4-ca. 300	32-34°S	St	South Africa	Kirkman <i>et al.</i> 2010
99	84	10-201	33°N	As	Japan	Kasuya 1986
20	8	1-100	34°S	As	South Africa	Best 2007
32	10-19	1-500	25-39°N	As	Japan	Miyashita 1993
58	n/a	1-ca. 300	33-45°S	St	New Zealand	Brabyn 1991

False killer whale social structure appears to show similarities to those of other globicephalids like pilot and killer whales (Connor *et al.* 1998). Larger groups may be comprised of smaller more stable subgroups (Reeves *et al.* 2002, Baird *et al.* 2010). Site fidelity, long-term associations between individuals and stable clusters with differing home ranges (Figure 1.5) have been recorded off Hawai'i (Baird *et al.* 2008, Baird *et al.* 2010, Baird *et al.* 2012), Costa Rica (Acevedo-Gutiérrez *et al.* 1997) and Gabon (Weir *et al.* 2013).

Satellite tagging of individuals in Hawaiian waters revealed that groups may disperse over > 25 km and still move in the same direction and engage in the same behaviour (Baird *et al.* 2008, Baird *et al.* 2010), and that individuals may separate for days and over > 100 km before re-associating (Baird *et al.* 2010). However, it is possible that the social structure observed in these genetically closed Hawaiian insular populations may not be representative of the species' overall social structure, a scenario evident in a number of delphinid species (*e.g.* Connor *et al.* 1998, Hoelzel *et al.* 2007). Indeed, there are some indications of differences in social structure, with both dispersal and non-dispersal from natal groups suggested for false killer whales from different regions. For example, Connor *et al.* (1998)

suggested possible natal philopatry based on the observed conspecific care in a stranding event in Florida. Conversely, Purves and Pilleri (1978) suggested that entire groups may consist of a single family, based on noticeable anatomical differences between individuals of different groups. However, males in the late maturing stages have been absent from groups involved in strandings in South Africa (Ferreira *et al.* 2014) and beach drives in Japan (Kasuya 1986), suggesting at least some degree of male dispersal in those regions, similar to that reported for sperm whales (*Physeter macrocephalus*) (Connor *et al.* 1998).

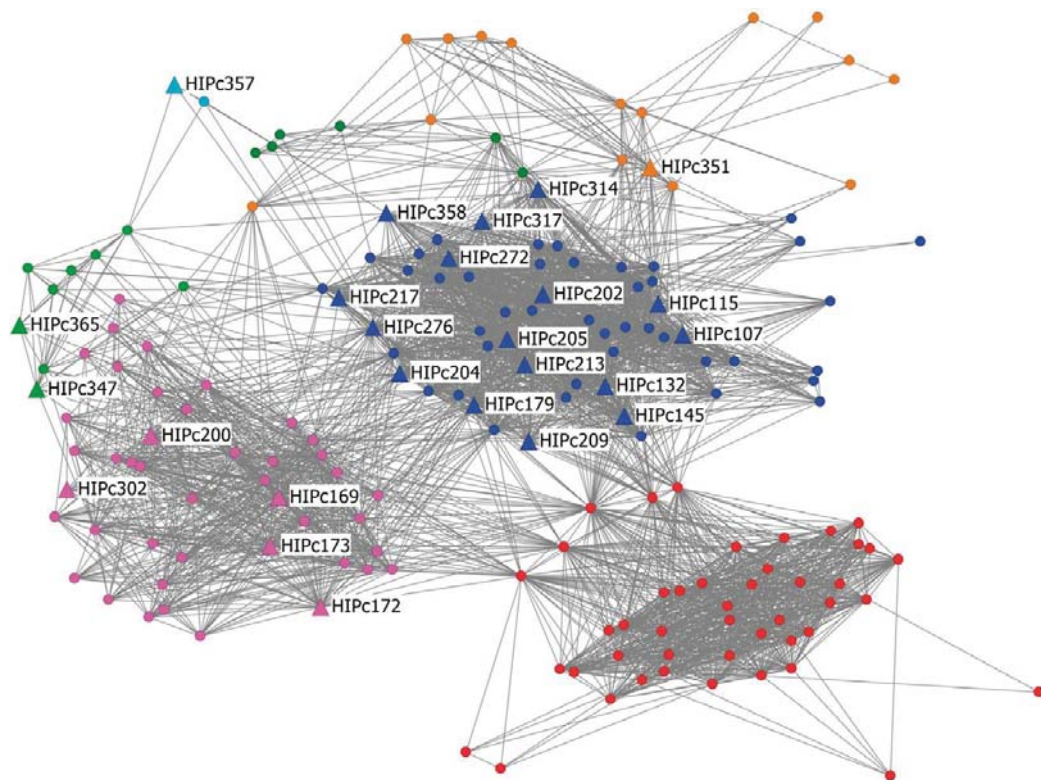


Figure 1.5. Example of false killer whale social structure of *distinctive* and *very distinctive* individuals from the Hawaiian insular population showing all members linked by association in a clustered social network. Source: Baird *et al.* 2012.

Virtually nothing is known about false killer whale social structure in New Zealand waters. Several large mass strandings, numbering up to 300 individuals per event (Brabyn 1991), indicate that social bonds between individuals also exist in the study area but beyond that, no other information on social structure in New Zealand waters could be found in the literature.

1.6.1.2 Interspecific associations

Interspecific associations between two or more cetacean species have been widely reported in the literature (e.g. Stensland *et al.* 2003). Like most globicephalids, false killer whales are known to regularly associate with a range of other cetaceans and have been observed in non-aggressive associations with a number of species. These include Risso's dolphin, pantropical spotted dolphin (*Stenella attenuata*), rough toothed dolphin (*Steno bredanensis*), melon-headed whale (*Peponocephala electra*), pygmy killer whale (*Feresa attenuata*), short-finned and long-finned pilot whale (*Globicephala melas*), although most frequently with the common bottlenose dolphin (*Tursiops truncatus*, hereafter referred to as bottlenose dolphin). A more detailed account of these interspecific associations is provided in Table 1.4. False killer whales have also been reported to strand together with other delphinids, including the bottlenose and Risso's dolphin (Best and Reeb 2010). False killer whales have produced viable hybrid offspring with bottlenose dolphins (Figure 1.6), at least in captivity (Nishiwaki & Tobayama 1982, Breese 1990). However, bottlenose dolphins are known to produce hybrid offspring with a range of other delphinids, including long-beaked common dolphin (*Delphinus capensis*, Zornetzer and Duffield 2003), rough-toothed dolphin (Dohl *et al.* 1974), Guiana dolphin (*Sotalia guianensis*, Caballero and Baker 2010), Risso's dolphin (Shimura *et al.* 1986) and the short-finned pilot whale (Sylvestre and Tanaka 1985). Despite being considered a relatively common occurrence (Reeves *et al.* 2002), the function or extent of these interspecific associations remain largely unknown.

Table 1.4. Cetacean species observed in non-aggressive interspecific associations with false killer whales.

Common name	Scientific name	Location	Source
Bottlenose dolphin	<i>Tursiops</i> spp.	Western tropical Pacific Eastern tropical Pacific Galapagos Islands Chile Japan China South Africa Maldives Gulf of Mexico New Zealand Indonesia Western tropical Pacific	Miyazaki and Wada 1978 Scott and Chivers 1990 Palacios 1996 Flores <i>et al.</i> 2003 Mizue and Yoshida 1961, Tsutsumi <i>et al.</i> 1961 Zhou <i>et al.</i> 1982 Best and Reeb 2010 Anderson 2005 Maze-Foley and Mulin 2006 Gaskin, 1972 Kreb and Budiono 2005 Miyazaki and Wada 1978
Long-finned pilot whale	<i>Globicephala melas</i>	Northern Indian Ocean	Leatherwood <i>et al.</i> 1984
Melon-headed whale	<i>Peponocephala electra</i>	Salomon Islands	Takekawa 2002
Pantropical spotted dolphin	<i>Stenella attenuata</i>	Hawaii Eastern tropical Pacific Northern Indian Ocean South Africa La Reunion Unspecified Hawaii	McSweeney <i>et al.</i> 2009 Scott and Chivers 1990 Leatherwood <i>et al.</i> 1984 Best and Reeb 2010 Poisson and Taquet 2001 Watson 1981 Baird <i>et al.</i> 2008
Pygmy killer whale	<i>Feresa attenuata</i>	Eastern tropical Pacific Chile	Scott and Chivers 1990, Flores <i>et al.</i> 2003
Risso's dolphin	<i>Grampus griseus</i>		
Short-finned pilot whale	<i>Globicephala macrorhynchus</i>		



Figure 1.6. The intergeneric hybrid between a false killer whale and a bottlenose dolphin (top), and its offspring (bottom), fathered by a bottlenose dolphin. Source: Sea Life Park, Hawaii.

Defran and Pryor (1980) propose that the formation of such mixed-species groups may result from shared or overlapping feeding grounds. Tsutsumi *et al.* (1961) also propose the availability of food resources as a possible reason behind these associations and further note that they only occur during winter months in Japanese waters, suggesting a seasonal aspect.

In addition to the frequent non-aggressive interspecific associations, aggressive associations with other cetacean species have also been observed. Harassment of bottlenose dolphins (Acevedo-Gutiérrez *et al.* 1997) and sperm whales, the latter whilst associating with bottlenose dolphins in a non-aggressive manner (Palacios and Mate 1996), have been reported. However it is difficult to accurately interpret such events. Additionally, false killer whales have been observed to prey on smaller delphinids during purse seine fishing operations (Perryman and Foster 1980), although this behaviour may at least in part have been influenced by the fishery. There is one report of predation on a humpback whale (*Megaptera novaeangliae*, Baird 2008) calf (see also section 1.7 on diet in this chapter).

1.6.2 Swimming and diving behaviour

Like other globicephalids (*e.g.* Heimlich-Boran 1988, Shane 1995), the species has been reported to travel in line-abreast formation, presumably to increase the likelihood of encountering prey (Alling *et al.* 1982). The mean fineness ratio of captive individuals has been calculated at 6.50 compared to 5.15 for bottlenose dolphin and 4.82 for killer whales (Fish 1998). An average travel speed of 15 kilometres per hour (km/h) was recorded during 16 hours of continuous observation off Chile (Sanino and Fowle 2006), while the horizontal movements of satellite tagged individuals in Hawaiian waters averaged 4.93 km/h, with movements of up to 18.6 km/h recorded. The maximum swim speeds of a captive specimen has been recorded at 26.9 km/h or 7.46 meters per second (m/s) (Fish 1998) and 28.8 km/h or 8.0 m/s (Rohr *et al.* 2002), respectively. These figures are very similar those recorded for common (8.0 m/s) and bottlenose dolphins (6.67-8.15 m/s, Rohr *et al.* 2002).

While Cummings and Fish (1971) estimated the maximum diving depth of false killer whales to be *ca.* 500 m, dives of up to 600 m have been documented for an individual in Japanese waters (Minamikawa *et al.* 2011) and 927.5 m recorded for one individual in Hawaiian waters (Baird *et al.* 2014). Dives of 14.5 and 15.2 minutes have been recorded in Japanese and Hawaiian waters respectively (Minamikawa *et al.* 2011, Baird *et al.* 2014). Both are shorter than the aerobic dive limit of 18.5 minutes, calculated by Minamikawa *et al.* (2011) who suggested that the discrepancy may be the result of rapid, and therefore more energy consuming, vertical movements observed during the study. Overall, these figures are comparable to the diving behaviour of other mid-size cetaceans such as long-finned pilot whales (*e.g.* Baird *et al.* 2002, Heide-Jørgensen *et al.* 2002).

1.6.3 Acoustics

False killer whales are considered an extremely vocal species (Murray *et al.* 1998), which has led to a significant number of acoustical studies, involving primarily captive specimens. The species produces a wide range of clicks, burst pulses and whistles as well as sounds that could be classed as intermediate between these categories (Murray *et al.* 1998). The dominant frequency of vocalizations by three captive individuals was 28 kHz, lasting 60 to 75 μ sec. and closely resembles the wave form of the bottlenose dolphin (Kamminga and van Velden 1987). Captive individuals have also produced high frequency vocalizations of *ca.* 100 kHz during echolocation experiments (Au *et al.* 1995). As with various other cetacean species (*e.g.* Whitehead 1998), intraspecific differences in vocalizations have been recorded. Higher averages of minimum and maximum frequency of vocalizations were recorded off Chile (Sanino and Fowle 2006), compared to individuals recorded in equatorial waters of the eastern tropical Pacific (Oswald *et al.* 2003). Similarly, a comparison of false killer whale vocalizations from the Caribbean and Costa Rica showed differences in pitch and the number of inflections, suggesting differences in vocalizations between regions and/or populations (Rendell *et al.* 1999).

1.6.4 Foraging behaviour

False killer whale foraging behaviour remains poorly documented, with only a small number of observations available in the literature. The species has been reported to feed both during the day and at night (Evans and Awbry 1988). However, decreased diving depth and lower and less variable swim speed at night may suggest a lower level of nocturnal feeding activity (Baird 2009) and/or foraging at shallow depths (Minamikawa *et al.* 2011). False killer whales have been observed to discard tails, gills and entrails of captured fish (Shallenberger 1981), and to share prey with conspecifics (Connor and Norris 1982, Rasmussen *et al.* 2002, Baird *et al.* 2008).

1.7 Diet

The diet of false killer whales appears to consist primarily of a variety of fishes and cephalopods (Stacey *et al.* 1994, Odell and McClune 1999), in particular large predatory fishes and epipelagic squids. Localised prey specialization in different populations has been suggested (Reeves *et al.* 2002, Ferreira 2008, Botta *et al.* 2011). Average daily food consumption of captive false killer whales was calculated to be between 3.4 and 4.3% of their body weight (Kastelein *et al.* 2000). A comprehensive overview of different prey species together with the locations where the observations were made is provided in Table 1.5.

Table 1.5. Annotated list of known false killer whale prey species. Note: Observed aggressive interactions with some cetacean species (*) may constitute harassment rather than predation.

Common name	Scientific name	Location	Source
Cephalopods			
Neon flying squid	<i>Ommastrephes bartramii</i>	Brazil	Andrade <i>et al.</i> 2001
Angolan squid	<i>Todarodes angolensis</i>	South Africa	Sekiguchi <i>et al.</i> 1992
Diamond squid	<i>Thysanoteuthis rhombus</i>	South Africa and Gran Canaria	Sekiguchi <i>et al.</i> 1992, Hernandez-Garcia 2002
	<i>Ommastrephes bartramii</i>	Gran Canaria	Hernandez-Garcia 2002
	<i>Argonauta</i> sp.	Gran Canaria	Hernandez-Garcia 2002
	<i>Chiroteuthis</i> sp.	Gran Canaria	Hernandez-Garcia 2002
Magister or Boreopacific armhook squid	<i>Berryteuthis magister</i> or <i>Gonatopsis borealis</i>	British Colombia	Baird <i>et al.</i> 1989
Unspecified squid		Brazil	Pinedo and Rosas 1989 in Botta <i>et al.</i> 2011
		Japan	Tsutsumi <i>et al.</i> 1961
Fishes			
Drums	Sciaenidae	Brazil	Pinedo and Rosas 1989 in Botta <i>et al.</i> 2011
Sea basses	Serranidae	Brazil	Pinedo and Rosas 1989 in Botta <i>et al.</i> 2011
Yellowtail	<i>Seriola</i> sp.	Japan	Kasuya 1985
Japanese sea bass	<i>lateolabrax japonicus</i>	Japan	Kasuya 1985
Jack mackerel	<i>Trachurus symmetricus</i>	Japan	Tsutsumi <i>et al.</i> 1961
Perch	<i>Lateolabrax japonicus</i>	Japan	Kasuya 1985
Salmon	<i>Oncorhynchus</i> sp.	British Colombia	Baird <i>et al.</i> 1989
Bonito	<i>Sarda lineolata</i>	California	Brown <i>et al.</i> 1966
Yellowfin tuna	<i>Thunnus albacares</i>	Hawaii	Baird <i>et al.</i> 2008
Albacore tuna	<i>Thunnus alalunga</i>	Hawaii	Baird <i>et al.</i> 2008
Skipjack tuna	<i>Katsuwonus pelamis</i>	Hawaii	Baird <i>et al.</i> 2008
Dolphin fish	<i>Coryphaena hippurus</i>	Hawaii	Baird <i>et al.</i> 2008
Wahoo	<i>Acanthocybium solandri</i>	Hawaii	Baird <i>et al.</i> 2008
Lustrous pomfret	<i>Eumegistus illustrus</i>	Hawaii	Baird <i>et al.</i> 2008
Scrawled file fish	<i>Aluterus scriptus</i>	Hawaii	Baird <i>et al.</i> 2008
Threadfin jack	<i>Alectis ciliaris</i>	Hawaii	Baird 2009
Broadbill swordfish	<i>Xiphias gladius</i>	Hawaii and unspecified locations	Baird <i>et al.</i> 2008, Ramos-Cartelle and Mejuto 2008
Roosterfish	<i>Nematistius pectoralis</i>	Costa Rica	Oviedo-Correa <i>et al.</i> 2009
Amberjack	<i>Seriola alandi</i>	Florida	Bullis and Moore 1956
Cod	(<i>Gadus callarias</i>)	Great Britain	Peacock <i>et al.</i> 1936 in Stacey <i>et al.</i> 1994
Catfish	<i>Tachysurus</i> sp.	Cape Comorin, Southern India	Silas and Kumara Pillay 1984
Elasmobranches			
Blue shark	<i>Prionace glauca</i>	Unspecified locations	Ramos-Cartelle and Mejuto 2008
Cetaceans			
Spotted dolphin	<i>Stenella</i> sp.	Eastern tropical Pacific	Perryman and Foster 1980
Spinner dolphin	<i>Stenella</i> sp.	Eastern tropical Pacific	Perryman and Foster 1980
Common dolphin	<i>Delphinus</i> sp.	Eastern tropical Pacific	Perryman and Foster 1980
Sperm whale*	<i>Physeter macrocephalus</i>	Galapagos Islands	Palacios and Mate 1996
Humpback whale*	<i>Megaptera novaeangliae</i>	Unspecified	Weller 2002, Baird 2008

False killer whales have also been observed to prey on other cetacean species such as pantropical spotted dolphins (*Stenella* sp.), spinner dolphins (*Stenella* sp.) and common dolphins (*Delphinus* sp.) in the eastern tropical Pacific, during purse-seine fishing operations (Perryman and Foster 1980). It has been suggested that such behaviour may at least in part be influenced by the anthropogenic impact of entrapping the respective dolphin species in

nets, allowing for easy access to an otherwise highly mobile and alert prey species (Leatherwood *et al.* 1989). However, there are further reports of harassment of and/or possible predation on large cetaceans, including the sperm whale (Palacios and Mate 1996) and the humpback whale (Weller 2002, Baird 2008), although it is unclear to what extent such events may constitute predation.

1.8 Conservation and management

1.8.1 Conservation status

The International Union for Conservation of Nature (IUCN) lists the false killer whale as *Data Deficient*, stating declining prey resources, low-level directed hunts, fishery interactions and climate change as the primary threats, albeit with unknown impact (IUCN 2013). Within New Zealand waters, the false killer whale is currently listed as *Not Threatened* (Baker *et al.* 2010). However the Department of Conservation concedes that *confidence in the listing is low due to there being only poor data available for assessment* (Townsend *et al.* 2008). Indeed, no species specific false killer whale research has been conducted in New Zealand, prior to the studies associated with the results presented herein.

1.8.2 Strandings

The biggest natural threat is likely the species' tendency to strand en masse, a phenomenon affecting most globicephalids, in particular pilot whales (Mitchell 1965, Caldwell *et al.* 1970, Odell *et al.* 1979, Baker 1981, Phillips 1988, Morimitsu *et al.* 1987, Alonso *et al.* 1999, Kirkman *et al.* 2010). As many as 835 individuals have been reported in a single stranding event (Ross 1984).

Mass strandings of false killer whales also occur in New Zealand, with both mass and single strandings reported (Brabyn 1991). To date, only 28 stranding events have been reported (New Zealand Stranding Database, Te Papa Tongarewa). Despite the infrequency of actual strandings, false killer whales are among the most numerous cetaceans to strand on New Zealand shores, second only to pilot whales (*Globicephala* spp.) (Baker 1981, Brabyn 1991), owing to the fact that false killer whale strandings constitute some of New Zealand's largest mass strandings, involving in one case *ca.* 300 animals during a single event (Brabyn 1991).

Investigations on the underlying causes of mass strandings often remain inconclusive and are the subject of continuing debate (*e.g.* Bradshaw *et al.* 2006). Sonar termination, caused by a gently sloping shore, and the presence of micro bubbles, interfering with echolocation, have been suggested as possible causes of the mass stranding of 120 false killer whales in

Western Australia in 2005 (Chambers and James 2005), while parasytogenic-induced echolocative dysfunction has been proposed as the cause of a mass stranding of 125 individuals in Japan in 1986 (Morimitsu *et al.* 1987). No forensic post-mortem examinations have been conducted on stranded false killer whales in New Zealand.

1.8.3 Fisheries interactions

Direct interactions with fisheries (the physical contact with fishing gear) have been described as the most pressing anthropogenic threat facing the world's marine mammals (Read 2008) and appear to be particularly relevant to false killer whales. In particular, the frequently reported depredation of long lines appears to be the most common and widespread source of conflict with fisheries, and has earned the false killer whale together with pilot whales the reputation of a 'problem species' for the industry at low latitudes of all three ocean basins (Bargain *et al.* 2000, Baird and Gorgone 2005, Gilman *et al.* 2006, Hernandez-Milian *et al.* 2008, Ramos-Cartelle and Mejuto 2008, Baird 2009).

The conflict is likely caused by the industry's targeting of and competing for known false killer whale prey species such as tuna (*Thunnus* spp.) and swordfish (*Xiphias gladius*). For false killer whales, entanglement in fishing gear and/or ingestion of hooks is thought to be the main risk resulting from such depredation attempts, known to cause injury and/or mortality (Forney and Kobayashi 2008, Ramos-Cartelle and Mejuto 2008).

False killer whale interactions with fisheries have been documented since at least the 1960s (Mizue and Yoshida 1961, Perryman and Foster 1980, Leatherwood *et al.* 1989, Odell and McClune 1999) and some of our knowledge about the species is the direct result of such interactions, in particular in Japanese waters (*e.g.* Ferreira *et al.* 2014, Kasuya 1986). Conflicts with Japanese fisheries, based on the perceived depletion of yellowtail (*Seriola quinqueradiata*) stocks and disturbance of fishing activity by false killer whales, have been the cause of directed and, in some cases, well documented shore drives (Kasuya 1985, Nishida and Yukiko 2002), which also target pilot whales and bottlenose and Risso's dolphins. Between 1965 and 1982, a minimum of 1141 false killer whales were harvested during such events in the Iki Island area alone (Kasuya 1985). As the records include hundreds of unidentified cetaceans (Kasuya 1985), the actual number of false killer whales taken is likely underestimated.

The species continues to be taken in Japanese waters, with the current annual quota set at 50 whales (including 40 individuals for beach drives and 10 individuals for harpoon fisheries, (Kasuya 2007), with a minimum of 269 individuals slaughtered between 1995 and 2004 (Kasuya 2007). While the effect of these takes on the local false killer whale population

remains unknown, a higher ovulation rate has been documented in individuals from that region, compared to specimens sampled from South African waters (Ferreira *et al.* 2014). Environmental factors, but also a possible response to exploitation, have been suggested as reasons behind these different ovulation rates (Ferreira 2008).

While interactions with fisheries may appear to have the most drastic effect on false killer whale populations in Japanese waters, there are indications that this impression may simply be the result of more accurate and publicly available reporting. For example, Yang *et al.* (1999) report that false killer whales are taken as bycatch in various fisheries in Taiwanese coastal waters, with annual numbers possibly ranging in the hundreds. False killer whales are also taken in the Hawaiian long line industry at an unsustainable rate (Carretta *et al.* 2007) (Figure 1.7), resulting in an apparent population decline among the insular population (Baird *et al.* 2005, Reeves *et al.* 2009). Low numbering accidental takes are also reported from coastal fisheries in Brazil (Reeves *et al.* 2003).



Figure 1.7. A false killer whale snagged on a fishing hook of the Hawaiian long line fishery © NMFS.

The sale of false killer whale meat at Korean markets, derived from individuals taken as bycatch, further shows that fishery interactions regularly have a lethal outcome for the species (Baker *et al.* 2006). While the impact of injuries and fatalities sustained during fishery interactions on false killer whale populations worldwide is difficult to assess, it

seems reasonable to assume that any false killer whale population in regular interaction with fisheries may be affected adversely.

1.8.4 Directed hunts

Although the shore drive operations carried out in Japan could be considered a directed hunt as much as a fishery interaction, false killer whales were never harvested by standard whaling operations in the traditional sense (Gaskin 1968). However, there is some evidence that the species, like many other small and medium sized cetacean species, is increasingly affected by unregulated whaling practices and/or targeted 'bycatch' (Reeves *et al.* 2003, Baker *et al.* 2006). For example, the amount of bycatch-derived (and mostly unreported) false killer whale meat offered on Korean markets, suggests, at the very least, an incentive for such bycatch to occur (Baker *et al.* 2006). False killer whales are also hunted at low levels in the Caribbean (Caldwell *et al.* 1970) and in Indonesian waters (Barnes 1991) and shore drives have also been reported from Taiwan (Abel and Leatherwood 1985).

1.8.5 Captivity

An increasing number of delphinaria and marine parks around the world has also seen a rise in the demand for false killer whales to be held in such facilities (Fisher and Reeves 2005, Zhang *et al.* 2012). Although a less common sight in marine parks, compared to the more popular bottlenose dolphins or killer whales, the species has been held in captivity around the world since at least 1963, including in the U.S.A (Brown *et al.* 1966), China (Zhang *et al.* 2012), Japan (Nishiwaki and Toboyama 1982, Kasuya *et al.* 1984), the Netherlands (Kastelein *et al.* 2000), Australia (Abel and Leatherwood 1985) and Indonesia (Fisher and Reeves 2005).

False killer whales have been successfully bred in captivity in a number of places (*e.g.* Nishiwaki and Toboyama 1982, Clark and Odell 1999) and the species has been described to be comparatively easy to train and to adapt quickly to captive conditions (Brown *et al.* 1966). However, as with other delphinids (Pryor 1973), aggressive behaviour towards humans and other marine mammals has also been reported in captivity (Defran and Pryor 1980). Live captures were originally sourced from the U.S.A. (taken from Californian and Hawaiian waters) (Brown *et al.* 1966, Shallenberger 1981), with some individuals also captured in Australian waters (Abel and Leatherwood 1985). From the 1980s onwards, wild caught individuals were increasingly sourced from shore drives in Japan (Figure 1.8) and Taiwan (Kasuya *et al.* 1984, Reeves *et al.* 1994). Between 1979 and 2002, at least 48 individuals were traded, the majority of which are thought to have been sourced from free-ranging populations (Fisher and Reeves 2005).



Figure 1.8. A false killer whale being selected by aquarium representatives at a drive hunt in Futo, Japan ©Sakae Hemmi/Elsa Nature Conservancy 2006.

1.8.6 Marine aquaculture

The global increase of marine aquaculture facilities may also affect false killer whales, with entanglement, reduced prey availability, habitat degradation and displacement from important parts of the species' home range suggested as associated risks for cetaceans (Kemper *et al.* 2003). Marine aquaculture facilities are known to adversely affect many cetacean species and/or populations frequenting coastal waters such as harbour porpoises (*Phocoena phocoena*), bottlenose dolphins or killer whales (Kemper *et al.* 2003).

While the predominant pelagic distribution of the false killer whale places it outside of most aquaculture zones, such facilities may affect populations that exhibit site-fidelity in coastal waters, as has been suggested in a case from Costa Rica (Oviedo-Correa *et al.* 2009). Furthermore, the projected global increase in offshore aquaculture ventures (Benetti *et al.* 2010) may constitute a future concern for open ocean species such as the false killer whale.

1.8.7 Pollution

False killer whales are also susceptible to a number of other anthropogenic threats such as pollution, with toxin levels of many sampled individuals comparable to those of killer whales (*e.g.* Ylitalo *et al.* 2009). Tissue analyses of individuals taken in Japanese drive fisheries revealed that false killer whales were the most contaminated of the nine species sampled, far exceeding the levels deemed safe for human consumption (Endo *et al.* 2005). Similarly, concentrations of organotin compounds in stranded individuals from Thailand

were the highest of the five cetacean species sampled in the study, with butyltin concentrations *ca.* 10 times higher than in the other species sampled (Harino *et al.* 2007).

Analysis of persistent organic pollutants measured in blubber samples of Hawaiian insular false killer whales, revealed levels high enough in some individuals to affect their health (Ylitalo *et al.* 2009, Foltz *et al.* 2014). Likewise, false killer whale tissue samples from Brazilian waters showed high levels of polybrominated diphenyl ethers (PBDEs) (Dorneles *et al.* 2010), and high levels in organochlorine compounds were also reported in stranded individuals from Vancouver Island, Canada (Jarman *et al.* 1996), and western and eastern Australia (Kemper *et al.* 1994). This suggests that high toxin levels affect the species on a broad scale. Their elevated trophic position (Baird 2008), make false killer whales highly susceptible to the accumulation of pollutants (Endo *et al.* 2005), although it is unknown to what extent they may contribute to immunosuppression (Baird 2008).

Ingestion of plastics and other marine debris is a growing conservation concern affecting many cetacean species, including the false killer whale (Simmonds 2012). While the extent and effect of plastic ingestion by false killer whales remains unknown, it has been suggested to cause impairment of the digestive process and even mortality in other cetacean species (Jacobsen *et al.* 2010). It has further been suggested that the transfer of toxins from ingested plastics into the tissues of animals, may pose a health risk for the affected individual (Teuten *et al.* 2009).

1.8.8 Other threats

Species misidentification, although not a threat in itself, has the potential to result in the inaccurate assessment of important conservation issues such as occurrence, abundance and the impact levels of certain threats. Uncertainties over correct identification have frequently led to deliberate pooling of globicephalids ('blackfish') at stranding events (*e.g.* Brabyn 1991) and during fishery interactions and/or drive hunts (*e.g.* Kasuya 1971) in the past. Due to the false killer whale's relative obscurity and resemblance to other globicephalids, the species continues to be misidentified in the field, even by experienced observers (*e.g.* Castro 2004, Baird 2010). Given its tendency to both, strand en masse and interact with fisheries, misidentification during such events has the potential to underestimate population declines (Figure 1.9).

False killer whales may also become adversely affected by the effects associated with climate change, such as ocean acidification (Lawler *et al.* 2007). The predicted increase in CO₂ levels has been suggested to severely affect the abundance of epipelagic squid (Fabry *et*

al. 2008), thought to be an important food source for false killer whales (Odell and McClune 1999).



Figure 1.9. A bycaught false killer whale being hauled aboard a purse seine fishing vessel in an unidentified location in the Pacific. The image is a still taken from a video produced by a nongovernmental organisation to raise awareness about mega fauna bycatch. The specimen was incorrectly identified as a pilot whale in the accompanying media release, highlighting the potential risk of under-reporting of fishery interactions for this species due to misidentification. Source: Greenpeace 2012

1.9 Thesis context and objectives

On a global scale, false killer whales remain one of the lesser-known cetaceans, with an even greater paucity of information on the species in New Zealand waters. Aside from a few mostly anecdotal reports, the only information published on the species in this region so far is associated with this study. Despite this lack of knowledge, false killer whales are the second most numerous species to strand on New Zealand shores. The prominent stranding history set against an almost complete absence of information about the species, highlights the need for a species-specific study. The aim of this thesis is to provide an initial assessment of the status of the false killer whale in New Zealand waters. As such it constitutes the first dedicated study of the species in the area and seeks to elucidate spatial, temporal and social aspects of false killer whale population ecology in the region.

The objectives of the present study are to assess the:

- Occurrence of false killer whales in New Zealand waters based on the spatial and temporal trends in sightings and stranding events.
- Population size, social organisation and site fidelity of New Zealand false killer whales based on photo-identification.
- Extent and function of false killer whale interspecific associations with other cetacean species.

This thesis is the result of the collation of governmental stranding records and sighting reports from whale-watch vessels and dedicated research platforms. Consequently, the study highlights both, the possibilities but also the limitations that such an opportunistic approach presents. Results presented in this thesis, and conclusions and recommendations made herein will be the first presented for false killer whales in the region, offering an important step towards greater understanding of this little known species. Aspects of this thesis have been published in peer reviewed publications (Visser *et al.* 2010, Zaeschmar *et al.* 2013, Zaeschmar *et al.* 2014, Appendices B, C and D) and are cited in the appropriate chapters. It is envisaged that this research will provide the necessary data to determine an accurate conservation status of false killer whales in New Zealand, allowing for more effective management.

Chapter Two

Occurrence in New Zealand waters



Stranded false killer whales at the Chatham Islands, March 2005 (Photo © Clinton Duffy).

2.1 Introduction

Knowledge of a species' occurrence and range elucidates an important aspect of its ecology and provides one of the basic elements required for effective management. The scarce and predominantly pelagic distribution of the false killer whale (*Pseudorca crassidens*) (Odell and McClune 1999) makes it a difficult species to study, resulting in a general lack of baseline data concerning its ecology and distribution. Stranding records and opportunistic sighting reports are, in many cases, the only available data to assess such a cryptic species' occurrence.

False killer whales are prone to strand en masse (Odell and McClune 1999), with much of the knowledge regarding the species' occurrence being derived from such events, including in New Zealand (Baker 1981, Brabyn 1991). The exact causes of strandings, especially mass stranding events, remain unclear and are the subject of continuing debate (Bradshaw *et al.* 2006). However, some broad trends are evident; mass strandings appear to be particularly frequent in oceanic species with high sociality such as sperm (*Physeter macrocephalus*), pilot (*Globicephala* spp.) and false killer whales (Oremus *et al.* 2013), and in certain locations known as so called stranding 'hot spots' or 'whale traps'. 'Whale traps' share a similar coastal topography, characterised by gently sloping sandy beaches with adjacent headlands. Additionally, they are, at times, subject to wind driven onshore currents (Brabyn 1991). However, the reasons why cetaceans strand in the first place remain poorly understood and many possible causes have been suggested over the years, including climatic and oceanographic factors (Evans *et al.* 2005), navigational errors caused by anomalies in the Earth's magnetic field (Klinowska 1986), large tidal movements (Best 1989), anthropogenic sound (Jepson *et al.* 2003), parasitic infestations (Morimitsu *et al.* 1987), instinctive land seeking drive, closed off ancient migratory routes (Brabyn 1991) or a response to high population density (Sergeant 1982).

While the underlying causes of strandings may be difficult to identify, records of such events are considered indicative of general cetacean frequency and distribution in a given area, and can be especially useful for those species rarely encountered at sea (Thompson *et al.* 2013). New Zealand is considered a global stranding 'hot spot' and it has been suggested that the region's high rate of strandings may be attributed to a relatively large number of so-called 'whale traps' (Brabyn and McLean 1992).

Our current knowledge of false killer whale occurrence in New Zealand waters is based mostly on a few published stranding records (Baker 1981, Brabyn 1991). Due to some observed similarities in stranding locations and the tendency to strand en masse, false killer

whales have been included in the category of so-called ‘offshore delphinids’ in previously published New Zealand stranding records (Brabyn 1991). The ‘offshore delphinids include: pilot whales, southern right whale dolphins (*Lissodelphis peronii*) and Risso’s dolphins (*Grampus griseus*). Additionally, there are some mostly anecdotal at-sea observations (e.g. Gaskin 1972). Available stranding and sighting records suggest that distribution may be wide, yet scarce, in the region. New Zealand is located at the juncture of subtropical current and subantarctic wind systems (Heath 1985), and while the average sea surface temperature of the whole region falls within false killer whales’ reported temperature range, only northern parts lie in the species’ preferred upper range (Chiswell 1994). The limits of false killer whales’ natural range could therefore, be assumed to lie within the New Zealand region.

The occurrence of cetaceans in a particular area is often related to the availability of food resources (Bluhn *et al.* 2008, Ainley *et al.* 2010). Likewise, it has been suggested that false killer whale occurrence in certain areas is linked to prey abundance (Tsutsumi *et al.* 1961). Consequently, the observation of foraging behaviour provides possible clues in regards to the species’ presence in a given location. Additionally, the identification of targeted prey species affords valuable insights into the’ position in the food web and the ecosystem services that may be provided.

In this chapter, a multi-disciplinary approach is applied to investigate false killer whale occurrence in New Zealand waters, providing the first insights into this infrequently encountered species (refer also to Zaeschmar *et al.* 2013, Appendix C and Zaeschmar *et al.* 2014, Appendix D). Stranding records of false killer whales are related to those of other cetacean species, in particular the so called ‘offshore delphinids’. Results are discussed in relation to presence/absence data and opportunistic sightings from whale-watch vessels to ascertain spatial and temporal occurrence. Seasonality and group size are investigated and compared to populations in other regions. The presence/absence of foraging behaviour is assessed to determine if false killer whales utilise the study area for feeding purposes. Finally, resulting conclusions and plausible hypotheses are presented.

2.2 Methods

2.2.1 Study Area

The surface waters of the New Zealand region are influenced by two main water masses; the subtropical waters to the north and the subantarctic waters to the south (Heath 1985) (Figure 2.1). The warm subtropical waters are high in salinity and are carried to New Zealand by the

East Auckland Current (EAUC), particularly influencing northeastern waters, with its associated East Cape Current (ECC), carrying warm subtropical waters as far as the Chatham Islands (Sutton and Roemmich 2001). The cold low salinity subantarctic waters are driven north by the West Wind Drift (WWD), with the associated northward flowing Southland and Westland currents carrying cool waters along both coasts of the South Island, extending as far as the Hawke's Bay region on the east coast and Cape Egmont on the west coast of the North Island.

The area where the two water masses meet is known as the Subtropical Convergence (SC). The position of the SC is variable, lying in the vicinity of the Fiordland region on the west coast, and ranging from the north of the South Island to the south of the North Island off eastern New Zealand, approximately following the 15°C isotherm in summer and the 10°C isotherm in winter. The SC constitutes a significant divide for many marine fauna, including cetaceans (Gaskin 1968).

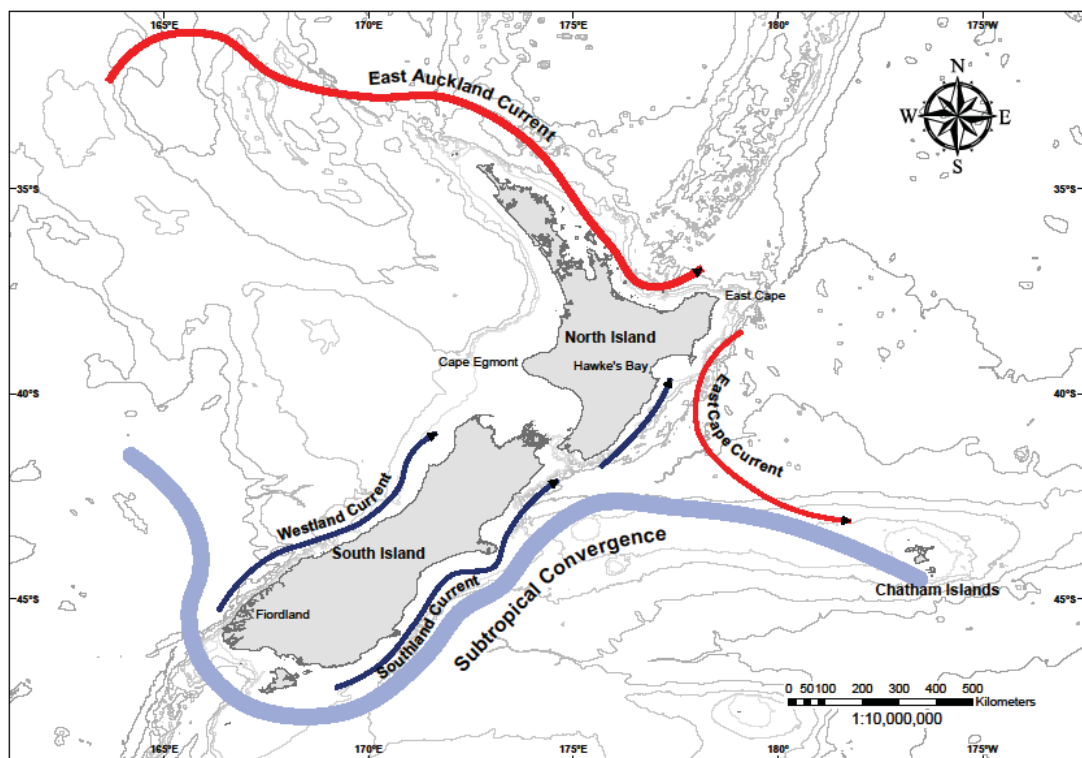


Figure 2.1. General water circulation around New Zealand. Warm and cold currents are shown in red and blue respectively. Note: The EAUC carrying warm water masses along the North Island's east coast and on towards the Chatham Islands via its associated East Cape Current (Based on Francis 1996).

2.2.1.1 Strandings

New Zealand has a prominent stranding history and is globally considered to be a stranding ‘hot spot’ for cetaceans in general, in particular in regard to mass strandings (Brabyn and McLean 1992). Within New Zealand, at least four general stranding ‘hot spots’ exist for all cetacean species: Whangarei and Hawke’s Bay on the North Island, Farewell Spit on the South Island and the Chatham Islands. When considering strandings of ‘offshore delphinids’, comprising of false killer whales, pilot whales, southern right whale dolphins and Risso’s dolphins, nine areas account for the vast majority of strandings, with five of those contained within the four larger ‘hot spots’ (Brabyn 1991). The other areas are located at Ninety Mile Beach, Doubtless Bay and the Coromandel Peninsula on the North Island and at Stewart Island to the south of the South Island (Brabyn 1991) (Figure 2.2).

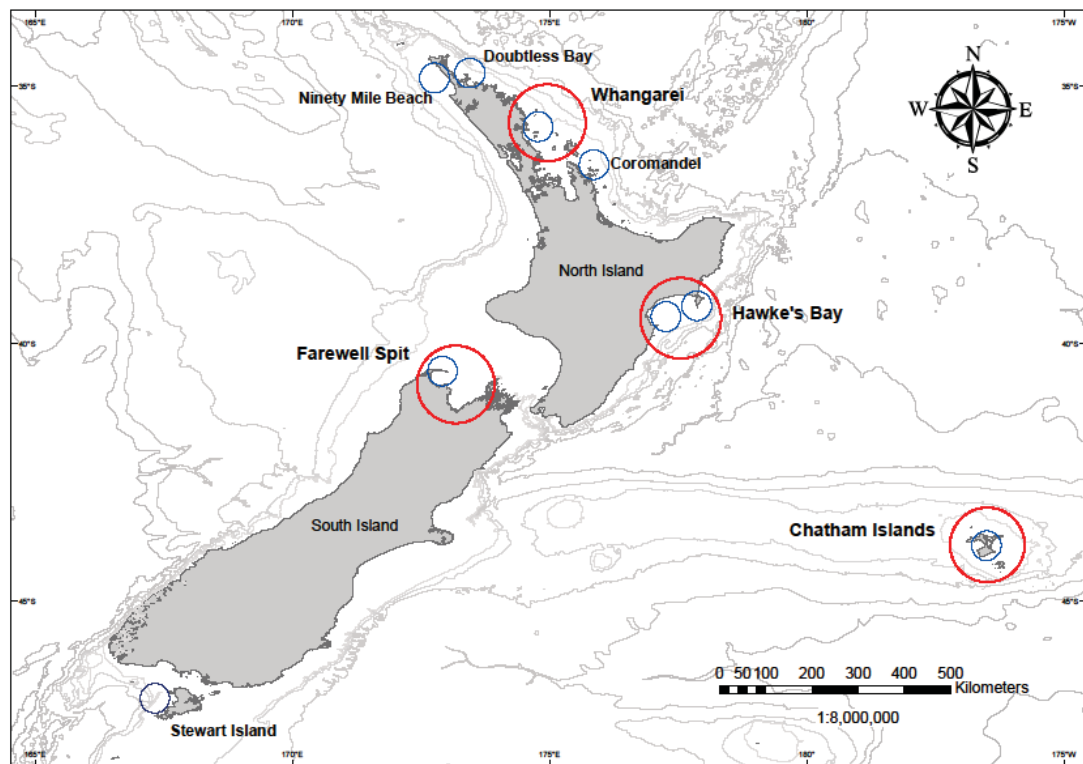


Figure 2.2. Map of New Zealand, showing the four most prominent mass stranding areas for cetaceans (large red circles) and the most prominent areas for strandings of offshore delphinids (small blue circles). The following species are classed as offshore delphinids: false killer whale, pilot whale, southern right whale dolphin and Risso’s dolphin. The majority of mass stranding events occur in these so called ‘whale traps’ (Source: Brabyn 1991).

2.2.1.2 At-sea observations

To maximise the chances of obtaining sighting information of the elusive false killer whales, the initial study area comprised the whole of New Zealand, based on the locations of whale- and dolphin-watch operators and dedicated cetacean research platforms that provided

information regarding the presence/absence of the species in their respective area of operation (Table 2.1, Figure 2.3). It is worth noting that there are no operators south of 38°S on the North Island's east coast or anywhere on the North Island's west coast or the South Islands' west coast north of Fiordland.

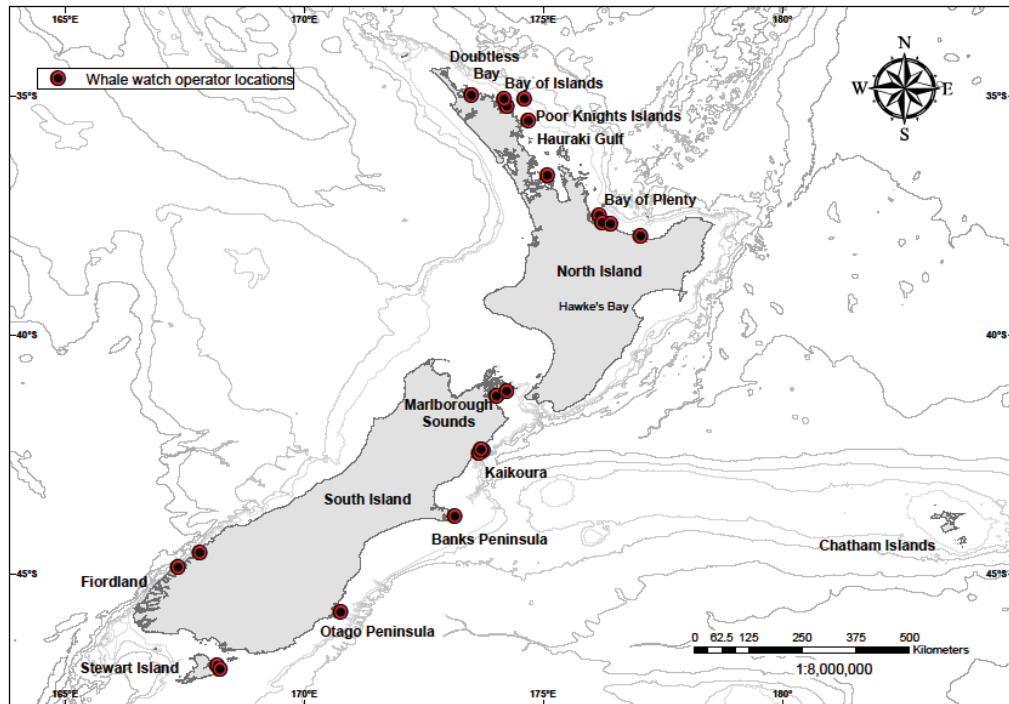


Figure 2.3. Locations of whale-watch operators and dedicated research platforms that provided presence/absence information of false killer whales in New Zealand waters between 1987 and 2014.

The assessment of the presence/absence reports enabled the identification of the principal focal area of this study: an approximate 650 kilometer (km) stretch of the northeastern coast of New Zealand, containing five locations where ongoing whale-watch operations and/or research projects were carried out. The study area extends from the Three Kings Islands (approximate position 34°09' S, 172°8' E) to East Cape (approximate position 37°05' S, 178°4' E, Figure 2.4). Hydrologically, the study area is influenced by the shoreward progression of the warm, southeastward flowing EAUC during December and its subsequent departure around May (Zeldis *et al.* 2004). The EAUC carries warm subtropical water (Sutton and Roemmich 2001) as well as associated marine fauna (Francis *et al.* 1999) into the study area. Sea surface temperature (SST) reaches 23° C during the austral summer and falls to 15° C in winter (Chiswell 1994). The austral seasons were defined as follows: spring (September – November), summer (December – February), autumn (March - May) and winter (June – August).

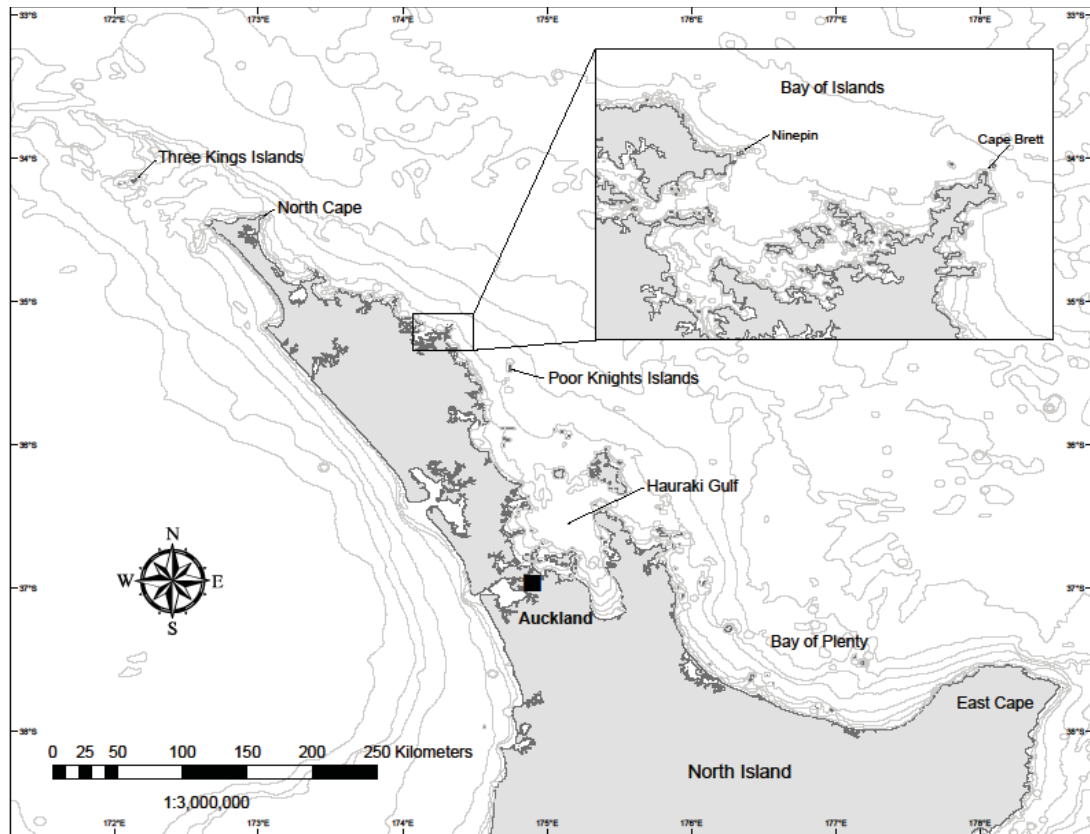


Figure 2.4. The study area with the five study locations along northeastern New Zealand. 1) Three Kings Islands, 2) Poor Knights Islands, 3) Hauraki Gulf, 4) Bay of Plenty and 5) Bay of Islands.

Sighting records of false killer whales were collected from five locations off northeastern New Zealand:

1: The Bay of Islands (BOI), (approximate position 35° S, 174° E). The area is characterised by a number of features; *ca.* 150 islands and islets with numerous bays and estuaries, and the Cape Brett peninsula, which intersects the warm EAUC, providing a large catchment area for nutrients (Baker and Madon, 2007). Water depth between the islands and the mainland is generally < 20 meter (m) while on the seaward side, water depth ranges generally from 50 to 120 m.

2: The Three Kings Islands (TKI), (approximate position 34°09' S, 172°8' E). Located 55 km north-west of New Zealand's North Island, situated near the continental shelf break, the area is characterised by submarine canyons and seamounts, with water depths generally ranging from 100 to > 1,600 m.

3: The Poor Knights Islands (PKI), (approximate position 35°28' S, 174°44' E). Located 19 km off the New Zealand's northeastern coast, these small islands (271 ha) are volcanic remnants that rise steeply from the otherwise flat ocean floor. Water depth in the area ranges from *ca.* 80 to > 150 m.

4: The Hauraki Gulf (HG), (approximate position 36°10'–37°10' S, 174°40'–175°30' E) is a shallow (< 60 m), semi-enclosed body of temperate water. Bathymetry in the area is predominantly flat.

5: The Bay of Plenty (BOP), (approximate position 36°30'–38°10' S, 175°40'–178°00' E) is a large open embayment with a generally flat seabed, containing a small number of islands. The shoreward flow of the EAUC ceases around East Cape, the southern limit of BOP, with the current turning away from the coast (Stanton *et al.* 1997). Water depth in the area ranges from 50 to > 200 m.

2.2.2 Data collection

2.2.2.1 Stranding records

Stranding records were obtained from Te Papa Tongarewa, the New Zealand Marine Mammal Stranding Database. The database is managed by the Department of Conservation (DOC) and has records dating back to 1840, which are verified and entered by DOC staff. The database was screened and any duplicate records or ambiguous entries removed. Species identification was based on available photographs, biopsy and skull samples and/or morphometric measurements. Additional records were included if they had been verified by Baker (1981) or Brabyn (1991) in former analyses using the New Zealand stranding records. Without these qualifiers, records were classed as *not verifiable*. Following Brabyn (1991), a single stranding was defined as one individual or a mother calf pair, while a mass stranding was defined as involving > 1 individual, except for mother/calf pairs.

The occurrence of strandings in relation to the southern oscillation index (SOI) was assessed to elucidate possible links between the El Niño Southern Oscillation (ENSO) and stranding events. The SOI provides an indication of the occurrence and intensity of El Niño and La Niña weather patterns, with sustained SOI of ≤ -8 or ≥ 8 considered indicative of El Niño and La Niña events, respectively. In the New Zealand region, El Niño events are associated with stronger westerly winds during summer and with stronger than usual southerly winds during winter. During La Niña episodes, increased SST and northeasterly winds are more prevalent (Goring and Bell 1999). SOI data were obtained from the Australian Bureau of Meteorology¹, which curates SOI records from 1876 to date. All verifiable recorded strandings since 1876 were analyzed.

¹The SOI archives can be accessed online. URL:
<http://www.bom.gov.au/climate/current/soihtm1.shtml>

2.2.2.2 *At-sea observations*

Presence/absence data from the records of whale- and dolphin-watch operators from around New Zealand and dedicated cetacean research platforms were used to provide an initial indication of false killer whale occurrence in the area. Twenty-one operators (11 from the South Island and 10 from the North Island) and two dedicated research platforms (both from the North Island) participated in a poll conducted between June and July 2014 and provided details on the existence or non-existence of false killer whale encounters in their respective areas of operation, taking into account seasonality, years of operation and the number of vessels (Table 2.1, refer to Appendix A).

Due to the heterogeneity of these data sources and the small sample size, a basic likelihood-of-encounter index was applied, based on the respective operator's area of operation in relation to false killer whales' known preference of warm open waters (Baird 2008). Consequently, false killer whales were presumed to be least likely encountered in enclosed inshore waters at higher latitudes and most likely in warmer open waters. Two equally weighted variables, (1) latitude and (2) proximity to shore were assessed. South Island waters were considered *high latitude* (high) and North Island waters were classified as *low latitude* (low). Vessels that operated primarily in enclosed waters (defined as harbours, embayments, fiords or sounds) were considered *inshore*, while those whose area of operation included open waters of depths > 50 m were considered *offshore*.

To avoid misidentification, especially confusion with the more common long-finned pilot whale (*G. melas*), a guide to correct identification was included (Figure 2.5). Positive sighting records were only classified as such if they were accompanied by photographs or video footage that confirmed species identity. Reports that could not be verified were classed as *not verifiable* and omitted from the analysis.

Table 2.1. Details of whale-watch operators and dedicated cetacean research platforms (*) that participated in the false killer whale sighting poll of false killer whales in their respective area of operation throughout the history of their operation between June and July 2014. South Island and North Island waters were classed as *high latitude* and *low latitude*, respectively. Vessels that operated primarily in enclosed waters were considered *inshore*, while those in open waters were considered *offshore*. Vessel types are aluminium (AL), steel (ST), timber (TIM), mono-hull (MON), catamaran (CAT) and helicopter (HEL).

North Island										
Name	Area of operation	Lat.	Oper. since	Season	Nr. of vessels	Vessel type	Length	Propulsion	Obs. eye height	Proximity
Cascade Charters	Three Kings Is.	33° S	2007-2008	Apr-May	1	FG, MON	16 m	Twin 435 hp	3 m	Offshore
Carino Dolphin Sail	Bay of Islands	35° S	1995	Sep-May	1	FG, CAT	14 m	Twin 40 hp	4 m	Inshore
Dolphin Discoveries	Bay of Islands	35° S	1992	All year	1	AL, CAT	18 m	Twin 375 hp	4 m	Offshore
Ecocruz	Bay of Islands	35° S	2000	Oct-May	1	ST, MON	22 m	120 hp	4 m	Offshore
Great Sights	Bay of Islands	35° S	1994	All year	2	FG, CAT	11 m	Twin 350 hp	3.5 m	Offshore
						AL, CAT	20 m	Twin 375 hp	4.5 m	
Dive Tutukaka	Poor Knights Is.	35° S	1999	All year	3	FG, MON	10-15 m	250 – 350 hp	3-4 m	Offshore
Ocean Blue	Poor Knights Is.	35° S	2006	All year	1	ST, MON	14 m	190 hp	3.5 m	Offshore
Dolphin Explorer	Hauraki Gulf	36° S	2002	All year	1	AL, CAT	20 m	Twin 350 hp	5 m	Inshore
Te Epiwhani*	Hauraki Gulf	36° S	2010-2012	All year	1	AL, MON	5.5 m	90 hp	2 m	Inshore
Dolphin Seafaris	Bay of Plenty	37° S	2006	Nov-May	1	FG, CAT	15 m	Twin 375 hp	4 m	Offshore
Aronui Moana*	Bay of Plenty	37° S	2010-2012	All year	1	AL, MON	5.5 m	90 hp	2 m	Inshore
Pee Jay White Is. Tours	Bay of Plenty	37° S	1995	All year	3	FG, CAT	22 m	Twin 750 hp	4 m	Offshore
South Island										
Name	Area of operation	Lat.	Oper. since	Season	Nr. of vessels	Vessel type	Length	Propulsion	Obs. eye height	Proximity
Dolphin Watch Tours	Marlborough Sds	41° S	1995	Oct-May	2	FG, CAT	9 - 13 m	Twin 250 – 350 hp	2 - 2.5 m	Inshore
Golden Future	Marlborough Sds	41° S	2002	All year	1	AL, MON	7.5 m	120 hp	2 m	Inshore
Dolphin Encounter	Kaikoura	42° S	1989	All year	3	FG, CAT	13 m	Twin 315 hp	4 m	Offshore
Whale Watch Kaikoura	Kaikoura	42° S	1987	All year	3	AL, CAT	18-20 m	Twin 375 – 450 hp	4 m	Offshore
World of Whales	Kaikoura	42° S	1990	All year	3	HEL.	-	-	-	Offshore
Black Cat Cruises	Banks Pen.	43° S	1988	All year	3	AL, MON	9 m	250 hp	2 m	Inshore
						AL, CAT	18 – 21 m	Twin 350-415 hp	4 m	
Real Journeys	Fiordland	44° S	2002	All year	4	AL, CAT	22 m	Twin 375 hp	4 m	Offshore
						ST, MON	30 m	475 hp	4.5 m	
						ST, MON	40 m	Twin 450 hp	7 m	
						ST, MON	42 m	Twin 475 hp	7 m	
Monarch Wildlife Cr.	Otago Pen.	45° S	2002	All year	1	TIM, MON	16 m	160 hp	4 m	Offshore
Fiordland Expeditions	Fiordland	45° S	2004	All year	2	ST, MON	17 – 20 m	135 – 195 hp	3 – 4 m	Offshore
Alhe Eco charters	Stewart Island	46° S	1999-2008	All year	1	AL, MON	7 m	125 hp	2 m	Offshore
Rakiura Charters	Stewart Island	46° S	2009	All year	2	AL, MON	8 m	150 hp	2 m	Offshore

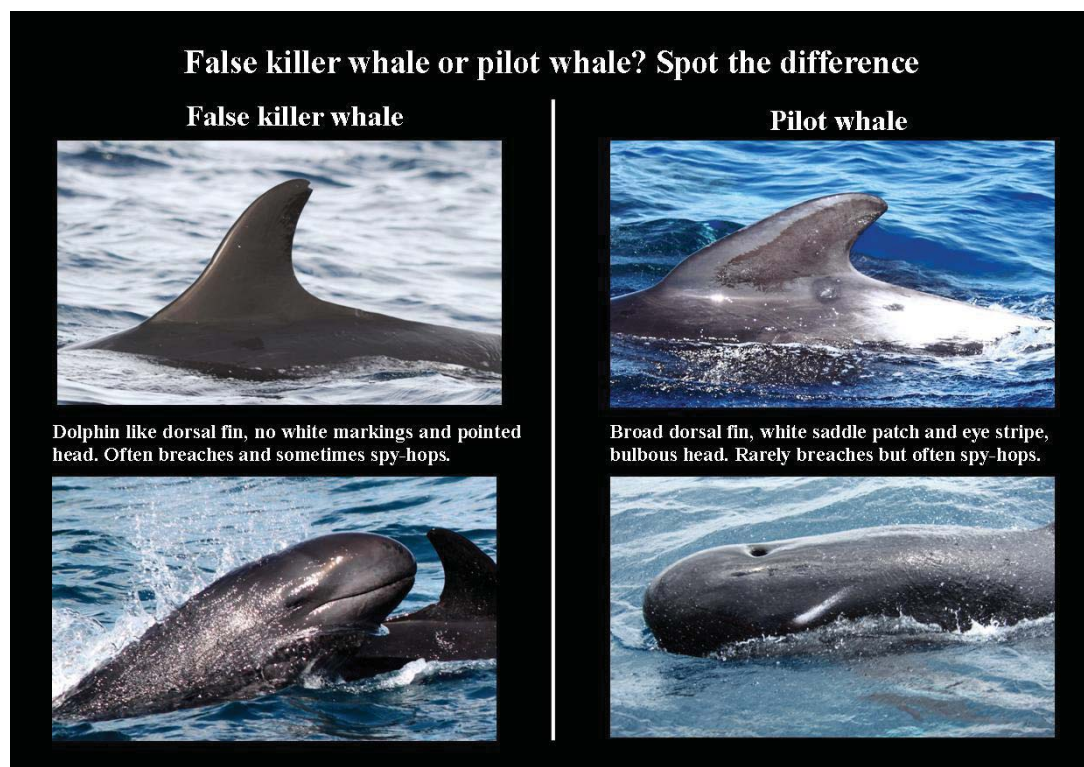


Figure 2.5. Identification guide for false killer whales highlighting the differences in appearance to long-finned pilot whales. The guide was included as part of a New Zealand-wide whale-watch operator poll between June and July 2014, investigating false killer whale sightings.

Tour vessels encountered false killer whales opportunistically during wildlife/marine tours throughout the study area, following a similar, asystematic survey methodology, which was dictated by factors such as prevailing weather conditions but also suspected areas of likely cetacean occurrence and/or sighting reports from other vessels or previous trips. Likewise, observer effort varied according to factors including the number of crew and passengers on board, weather conditions and previous sightings.

The research vessels encountered false killer whales during dedicated cetacean surveys undertaken when visibility was $> 1\text{km}$ and sea-state was \leq Beaufort Sea State (BSS) 3. Research vessels employed a continuous scanning methodology (e.g. Mann 1999), with trained observers using both naked eye and binoculars. False killer whales were detected by sighting cues such as splashes, silhouettes of surfacing animals and the presence of foraging associated species, in particular shearwaters (*Puffinus* spp.), which are known to associate with the species (Pitman and Balance 1992). Once detected, focal groups were approached to *ca.* 300 m and the vessels were slowed down to $\leq 8\text{ km/h}$. At this point, the encounter time and the vessels' initial GPS coordinates and depth were noted, together with environmental information such as BSS, wind speed and direction estimates, and SST readings using onboard thermometers. The focal group was then approached slowly, with

the vessels moving line abreast to travelling groups or slowly approaching stationary groups from the side and slightly behind, avoiding sudden changes in boat speed or direction of travel.

Ad libitum behavioural observations (Altmann 1974), focusing only on the presence or absence of foraging behaviour were recorded in transcript, audio log (dictaphone), or video form. Due to the ambiguity of cetacean behaviour states and the need for experienced observers to accurately identify them, only foraging observations were included in the analysis. This state was considered more reliably assessed by observers and could be more readily identified from photographs. Following Acevedo-Gutiérrez *et al.* (1997), foraging was defined by frequent, asynchronous dives with subgroups dispersing over larger areas, as well as by the presence of fish near the surface or sea birds feeding on fish remains near the whales. Prey species were identified from direct observations or photographs where possible.

Following Baird *et al.* (2008), it was assumed that all false killer whales occurring in the area at any one time were part of the same group. These parameters are considerably broader than those conventionally applied to other delphinid groups. However, it is supported by the infrequency of false killer whale encounters in the study area and the fact that false killer whales may at times disperse > 20 km and still be moving in the same direction and engaged in the same behaviour (Baird *et al.* 2008). Consequently, no spatial parameters were placed on group definitions. Given the frequency of association between false killer whales and common bottlenose dolphins (presumed to be of the offshore form and hereafter referred to as bottlenose dolphins), for the purpose of this study, the term mixed-species group refers to associations between these two species only. Following Shane (1990), a mixed-species group was defined as any number of individuals of one species observed in apparent association with the other species and generally moving in the same direction and engaged in similar behaviour.

As the majority of observations described herein were of large, dispersed groups containing more than one cetacean species and recorded by several observers, group size estimates may be biased, with results suggesting that they may be overestimated. Consequently, records, which only provide one group size estimate, instead of the minimum, best and maximum, are treated as the maximum group size estimates. For those records that differentiate, the maximum estimate was used for consistency. Additionally, the total numbers of photo-identified individuals were used to validate group size. Photo-identification methods are described in Chapter Three.

2.2.3 Analysis

Seasonality of strandings and the relation to SOI values was assessed using a Kolmogorov-Smirnov test in Minitab 17². As distribution failed normality criteria, the non-parametric Kruskal-Wallis test was applied. Despite apparent trends, no significant differences between stranding months or SOI values were found ($p = 0.968$). Consequently, no further categorization of stranding records was tested.

Sighting data from the operators and research platforms were filtered to ensure reliability. For the assessment of general occurrence, only records supported by proof of the species' identity (via photographs or video footage) and correct date (at least to the month) were included. For effort based analysis, a number of criteria had to be met: (1) year round operation, (2) complete and detailed records available (3) all records verifiable and (4) a significant number of records. As most of the observation platforms did not operate in the respective locations on a year round basis, had insignificant number of sighting records and/or could not provide exact numbers of sighting records, seasonal occurrence was assessed only from the records of the Great Sights vessel *Tutunui*, which kept reliable records and ran continuous trips up to twice daily in BOI between 1995 and 2007. Following Wiseman *et al.* (2011), a monthly index of false killer whale encounters was determined using a trip encounter rate (TER), which was calculated from the number of trips on which whales were encountered in proportion to the total number of trips undertaken that month. Monthly results across different years were pooled and calculated as an average sighting rate per 100 trips. To increase sample size, sightings were further pooled into warm (December – May) and cold (June – November) seasons. False killer whale occurrence in relation to SST was analysed.

Sightings in relation to depth and/or distance from shore were not tested as these were considered a possible function of limited search effort within inshore waters and not reflective of false killer whales' predominant deep water distribution. Consequently, results regarding depth and distance from shore are only included herein to illustrate that false killer whales do venture into shallow coastal waters in the study area. Occurrence of sightings in relation to ENSO was assessed from SOI records of the Australian Bureau of Meteorology and BOI false killer whale sighting records of *Tutunui* (1995 – 2007). To test for normality, a Kolmogorov-Smirnov test was used using Minitab 17 and a non-parametric Kruskal-Wallis test was selected. Significance levels were set at 0.05. To test for the significance of encounter duration and presence/absence of foraging on group size estimates, a General

² Minitab Inc. (2010). URL: www.minitab.com.

Linear Model (GLM) with a negative binomial model approach was applied. All analyses were completed in R using the “stats” (R Core Team³) and “Mass” (Venables and Ripley 2012) packages.

2.3 Results

2.3.1 Strandings

A total of 29 false killer whale stranding events have been documented in New Zealand since 1870, averaging one stranding event every *ca.* 5 years (Table 2.2). Of these, 3.4% ($n = 1$) was found to be a duplicate record and omitted, resulting in $n = 28$ *verifiable* events. Despite the low frequency in strandings, the total number of individuals involved in such events was high ($n = 735$), due to two particularly large events in 1943 and 1978, involving 253 and 300 individuals respectively and accounting for 75.2% of the total number of stranded individuals. False killer whales were the second most numerous cetacean to strand on New Zealand shores after pilot whales (Table 2.3). However, in terms of the number of actual stranding events, the species ranked 20th out of 47 (or 19th out of 45 if omitting the 3 ‘unidentified’ cetacean categories, Table 2.4).

As distribution of stranding data failed normality criteria, (Kolmogorov-Smirnov $Z = 0.212$, $SD = 1.614$, $p = 0.138$), the non-parametric Kruskal-Wallis test was applied. Despite apparent trends, no significant differences between month and strandings were found. Consequently, no further categorization of stranding records was tested. As these results are most likely a function of small sample size, trends instead of statistical significance will be discussed.

³ R Core Team, 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: <http://www.R-project.org/>

Table 2.2. Twenty eight false killer whale strandings recorded in New Zealand since 1870, as archived in the New Zealand Marine Mammal Stranding Database, Te Papa Tongarewa. Qualifiers for species verification are previously published record (PB), photograph (P), biopsy sample (B), skull sample (S) and morphometric measurements (M). Coasts are North Island east coast (NI-E), North Island west coast (NI-W), South Islands east coast (SI-E), South Island west coast (SI-W) and Chatham Islands (CH-I).

Date	Location	Coast	Group size	Verifiable	Qualifier	Notes
Jan. 18, 1870	Wellington, Lyall Bay	NI-W	1	Y	PB	
Mar. 1, 1906	Chatham Islands	CH-I	14	Y	PB	
Aug. 8, 1925	Wellington, Sinclair Hd.	NI-W	1	Y	PB	5 m male
Jan. 1, 1932	Chatham Islands	CH-I	1	Y	PB	
May 1, 1933	Gisborne, Awatere River	NI-E	1	Y	PB	
Sep. 8, 1935	Mahia Peninsula, Opoutama Beach	NI-E	38	Y	PB	
Jun. 17, 1936	Hawkes Bay, Napier	NI-E	29	Y	PB	Males and females
Apr. 8, 1943	Mahia Peninsula, Opoutama Beach	NI-E	300	Y	PB	Also some <i>Globicephala</i> sp.
Jul. 23, 1962	Otago, Warrington	NI-E	19	Y	PB	Males and females
Oct. 1, 1968	Hawkes Bay, Napier	NI-E	1	Y	PB	11ft
Jun. 9, 1969	Mahia Peninsula	NI-E	7	Y	PB, S	Together with 3 pygmy sperm whales (<i>Kogia breviceps</i>)
Jan. 1, 1970	Manawatu Beach	NI-E	1	Y	PB	
Apr. 1, 1970	Mahia Peninsula	NI-E	1	Y	PB	13 ft 11"
Apr. 7, 1970	Mahia Peninsula	NI-E	2	Y	PB	13 ft female and calf
Jun. 1, 1970	Mahia Peninsula, Opoutama Beach	NI-E	1	Y	PB	4.98 m very decomposed
Sep. 6, 1970	Mahia Peninsula, Opoutama Beach	NI-E	1	Y	PB	16'6"
Dec. 1, 1976	Northland, Ninety Mile Beach	NI-W	1	Y	PB	8' approx.
Mar. 31, 1978	Manukau Harbour	NI-W	253	Y	PB	
Feb. 20, 1980	Chatham Islands	CH-I	1	Y	PB, P	2 m
Feb. 23, 1980	Chatham Islands	CH-I	1	Y	PB, P	4 m. Group of 16 whales stayed in a very confined area of the harbour for 6-7 days. Left on February 25.
May 14, 1984	Westland, South Okarito Beach	SE-W	1	Y	PB, P, S	Female, 5.8 m
Mar. 20, 1988	Waikato, Taharoa beach	NI-W	1	Y	PB, P, M	Female, 3.73 m
Apr. 3, 1992	Taranaki, Opunake	NI-W	1	Y	PB, P, M	Female, 3.2 m
Feb. 7, 1993	Hawkes Bay, Ongaonga	NI-E	1	Y	P, M	Male
Mar. 12, 1993	Hawkes Bay, Gisbourne	NI-E	3	Y	P, B, M	1 Male, 3.9m, 1 female, 3.1 m, 1 female 2.2 m
May 15, 1998	North Auckland, Poutu Peninsula	NI-W	1	Y	P	
Apr. 13, 2000	Gisbourne, Anaura bay	NI-E	1	Y	P, B, M	Female, 2.89 m
Mar. 1, 2005	Chatham Islands	CH-I	53	Y	P, B, M	

Table 2.3. List of cetacean species stranded in New Zealand between 1840 and 2014 ordered by the total number of individuals involved. Records for long-finned, short-finned and unidentified pilot whale were pooled into one category (Pilot whale, *Globicephala* spp.). (Source: New Zealand Marine Mammal Stranding Database, Te Papa Tongarewa).

Species		No. incidents	No. individuals
Pilot whale	<i>Globicephala</i> spp.	370	12556
False killer whale	<i>Pseudorca crassidens</i>	27	735
Common dolphin	<i>Delphinus</i> sp.	454	657
Sperm whale	<i>Physeter macrocephalus</i>	241	525
Pygmy sperm whale	<i>Kogia breviceps</i>	391	485
Gray's beaked whale	<i>Mesoplodon grayi</i>	287	466
Hector's dolphin	<i>Cephalorhynchus hectori</i>	351	363
Bottlenose dolphin	<i>Tursiops truncatus</i>	153	318
Dusky dolphin	<i>Lagenorhynchus obscurus</i>	108	143
Cuvier's beaked whale	<i>Ziphius cavirostris</i>	119	123
Strap toothed whale	<i>Mesoplodon layardii</i>	99	113
Southern right whale dolphin	<i>Lissodelphis peronii</i>	17	103
Killer whale	<i>Orcinus orca</i>	50	92
Common minke whale	<i>Balaenoptera acutorostrata</i>	88	88
Unidentified beaked whale	<i>Mesoplodon</i> sp.	63	76
Pygmy right whale	<i>Caperea marginata</i>	62	63
Unidentified <i>Stenella</i>	<i>Stenella</i> sp.	39	56
Arnoux's beaked whale	<i>Berardius arnouxii</i>	43	54
Unidentified cetacean		43	46
Maui's dolphin	<i>Cephalorhynchus hectori maui</i>	40	42
Southern bottlenose whale	<i>Hyperoodon planifrons</i>	27	34
Andrew's beaked whale	<i>Mesoplodon bowdoini</i>	25	30
Shepherd's beaked whale	<i>Tasmacetus shepherdi</i>	28	29
Bryde's whale	<i>Balaenoptera edeni</i>	23	24
Antarctic minke whale	<i>Balaenoptera bonaerensis</i>	21	21
Humpback whale	<i>Megaptera novaeangliae</i>	21	21
Unidentified baleen whale	<i>Balaenoptera</i> sp.	20	20
Risso's dolphin	<i>Grampus griseus</i>	12	18
Fin whale	<i>Balaenoptera physalus</i>	15	15
Hector's beaked whale	<i>Mesoplodon hectori</i>	14	15
Blue whale	<i>Balaenoptera musculus</i>	14	14
Striped dolphin	<i>Stenella coeruleoalba</i>	13	14
Sei whale	<i>Balaenoptera borealis</i>	11	11
Dwarf sperm whale	<i>Kogia sima</i>	4	6
Rough toothed dolphin	<i>Steno bredanensis</i>	2	5
Pygmy blue whale	<i>Balaenoptera musculus brevicau</i>	4	4
Ginkgo toothed beaked whale	<i>Mesoplodon ginkgodens</i>	4	4
Unidentified toothed whale		4	4
Southern right whale	<i>Eubalaena australis</i>	3	3
Blainville's beaked whale	<i>Mesoplodon densirostris</i>	3	3
Spectacled porpoise	<i>Phocoena dioptrica</i>	3	3
Hourglass dolphin	<i>Lagenorhynchus cruciger</i>	2	2
Spade toothed whale	<i>Mesoplodon traversii</i>	1	2
Pygmy killer whale	<i>Feresa attenuata</i>	1	1
True's beaked whale	<i>Mesoplodon mirus</i>	1	1
Melon headed whale	<i>Peponocephala electra</i>	1	1
Pantropical spotted dolphin	<i>Stenella attenuata</i>	1	1

Table 2.4. List of cetacean species stranded in New Zealand between 1840 and 2014 ordered by the total number of discrete events. Records for long-finned, short-finned and unidentified pilot whale were pooled into one category (Pilot whale, *Globicephala* spp). (Source: New Zealand Marine Mammal Stranding Database, (Te Papa Tongarewa).

Species		No. incidents	No. individuals
Common dolphin	<i>Delphinus</i> sp.	454	657
Pygmy sperm whale	<i>Kogia breviceps</i>	391	485
Pilot whale	<i>Globicephala</i> spp.	370	12556
Hector's dolphin	<i>Cephalorhynchus hectori</i>	351	363
Gray's beaked whale	<i>Mesoplodon grayi</i>	287	466
Sperm whale	<i>Physeter macrocephalus</i>	241	525
Bottlenose dolphin	<i>Tursiops truncatus</i>	153	318
Cuvier's beaked whale	<i>Ziphius cavirostris</i>	119	123
Dusky dolphin	<i>Lagenorhynchus obscurus</i>	108	143
Strap toothed whale	<i>Mesoplodon layardii</i>	99	113
Common minke whale	<i>Balaenoptera acutorostrata</i>	88	88
Unidentified beaked whale	<i>Mesoplodon</i> sp.	63	76
Pygmy right whale	<i>Caperea marginata</i>	62	63
Killer whale	<i>Orcinus orca</i>	50	92
Arnoux's beaked whale	<i>Berardius arnouxii</i>	43	54
Unidentified cetacean		43	46
Maui's dolphin	<i>Cephalorhynchus hectori maui</i>	40	42
Unidentified <i>Stenella</i>	<i>Stenella</i> sp.	39	56
Shepherd's beaked whale	<i>Tasmacetus shepherdi</i>	28	29
False killer whale	<i>Pseudorca crassidens</i>	27	735
Southern bottlenose whale	<i>Hyperoodon planifrons</i>	27	34
Andrew's beaked whale	<i>Mesoplodon bowdoini</i>	25	30
Bryde's whale	<i>Balaenoptera edeni</i>	23	24
Antarctic minke whale	<i>Balaenoptera bonaerensis</i>	21	21
Humpback whale	<i>Megaptera novaeangliae</i>	21	21
Unidentified baleen whale	<i>Balaenoptera</i> sp.	20	20
Southern right whale dolphin	<i>Lissodelphis peronii</i>	17	103
Fin whale	<i>Balaenoptera physalus</i>	15	15
Hector's beaked whale	<i>Mesoplodon hectori</i>	14	15
Blue whale	<i>Balaenoptera musculus</i>	14	14
Striped dolphin	<i>Stenella coeruleoalba</i>	13	14
Risso's dolphin	<i>Grampus griseus</i>	12	18
Sei whale	<i>Balaenoptera borealis</i>	11	11
Dwarf sperm whale	<i>Kogia sima</i>	4	6
Pygmy blue whale	<i>Balaenoptera musculus breviceps</i>	4	4
Ginkgo toothed beaked whale	<i>Mesoplodon ginkgodens</i>	4	4
Unidentified toothed whale		4	4
Southern right whale	<i>Eubalaena australis</i>	3	3
Blainville's beaked whale	<i>Mesoplodon densirostris</i>	3	3
Spectacled porpoise	<i>Phocoena dioptrica</i>	3	3
Rough toothed dolphin	<i>Steno bredanensis</i>	2	5
Hourglass dolphin	<i>Lagenorhynchus cruciger</i>	2	2
Spade toothed whale	<i>Mesoplodon traversii</i>	1	2
Pygmy killer whale	<i>Feresa attenuata</i>	1	1
True's beaked whale	<i>Mesoplodon mirus</i>	1	1
Melon headed whale	<i>Peponocephala electra</i>	1	1
Pantropical spotted dolphin	<i>Stenella attenuata</i>	1	1

2.3.1.1 Stranding locations

The records analysed herein reveal false killer whale stranding events from the east and west coasts of both, the North and the South Island, spanning from Ninety Mile beach in the far northwest (approx. position 34°46' S, 172°58' E) to Warrington, Otago in the far southeast (approx. position 45°04' S, 170°36' E) (Figures 2.6 & 2.7). The majority of documented strandings occurred on the North Island (75.0%, $n = 21$), with the east coast showing a particularly high proportion of all strandings (64.3%, $n = 18$) and 85.7% of North Island strandings ($n = 18$).

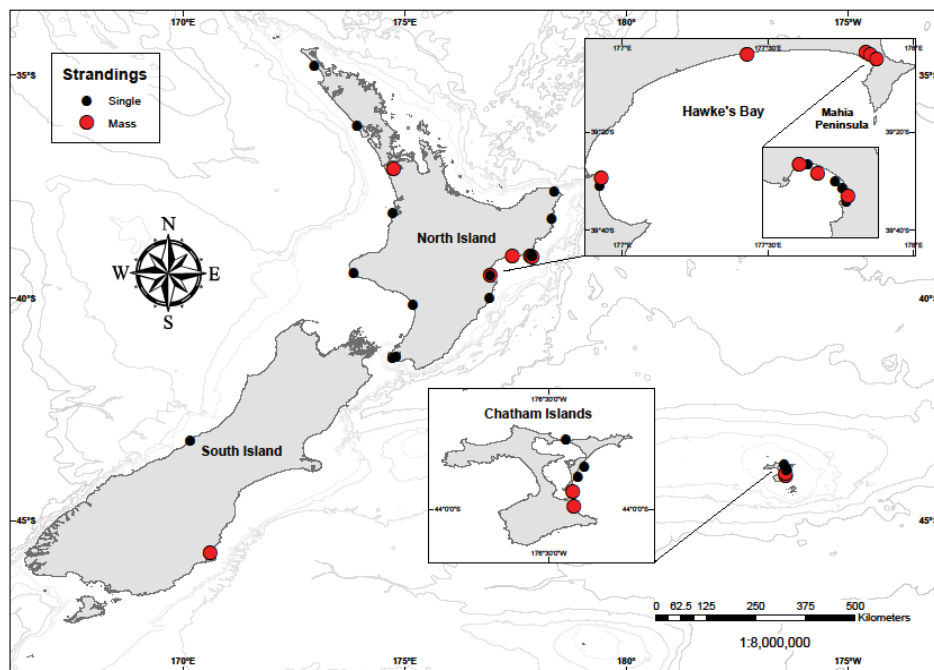


Figure 2.6. Locations of false killer whale strandings in New Zealand, documented since 1870, with high concentrations of strandings evident in the Hawkes Bay area and in the Chatham Islands. Mass strandings are represented by large red circles and single strandings are represented by small black circles.

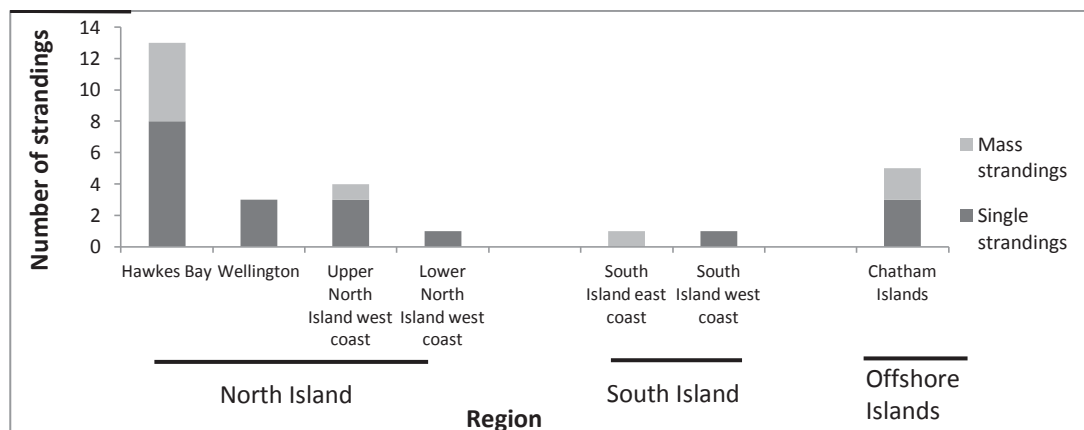


Figure 2.7. Locations of the 28 documented false killer whale strandings since 1870 in New Zealand.

Along the North Island's east coast, the Hawkes Bay region accounted for 46.4% ($n = 13$) of all reported strandings, 61.9% ($n = 13$) of North Island strandings ($n = 21$) and 72.2% ($n = 13$) of North Island east coast strandings ($n = 18$). The *ca.* 5 km long Mahia Beach on Mahia Peninsula constitutes the most prominent false killer whale stranding location in the New Zealand records (25.0% of all strandings, 33.3% of mass-strandings, 33.3% of North Island strandings and 38.8% of North Island east coast strandings, $n = 7$), followed by the Chatham Islands (17.6% of all strandings, $n = 5$). Only two stranding records exist from the South Island, with one documented stranding each from the east and west coasts (7.1% respectively).

Using single-dead strandings ($n = 18$) as an indicator of natural distribution (Brabyn 1991), 77.8% ($n = 14$) occurred on North Island shore, followed by 16.7% ($n = 3$) on the Chatham Islands and 5.6% ($n = 1$) on the South Island (Westland). Of the 14 North Island records, 71.4% ($n = 10$) were reported from the east coast, compared to 28.6% ($n = 4$) on the west coast. The Hawkes Bay region was again the most prominent location, accounting for 30.2% ($n = 7$), of all single dead strandings New Zealand-wide, 50% on the North Island and 70% of North Island east coast strandings.

Focusing only on the nine documented mass strandings, 77.7% occurred in just two areas; Hawkes Bay (55.5%, $n = 5$) and the Chatham Islands (22.2%, $n = 2$). The remaining two records were reported from the North Island's upper west coast and the South Island's lower east coast.

2.3.1.2 Stranding seasonality

While stranding reports exist for all months except November, seasonality was apparent, with 75.0% of incidents ($n = 21$) observed between January and June (Figure 2.8). Austral autumn was the most prominent stranding season (46.4%, $n = 13$), followed by summer (25.0%, $n = 7$). Winter and spring accounted for 17.9% ($n = 5$) and 10.7% ($n = 3$) of strandings, respectively.

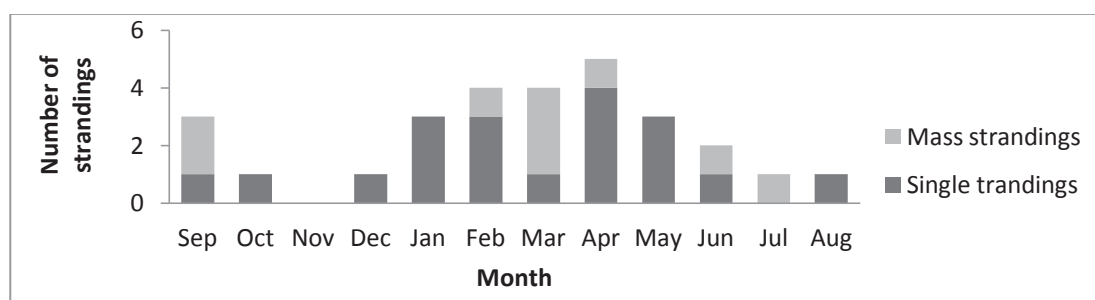


Figure 2.8. Seasonality of the 28 documented false killer whale strandings in New Zealand since 1870.

The most common stranding month was March (17.9%, $n = 5$), followed by February and April (each 14.3%, $n = 4$), and September (10.7%, $n = 3$). Single-dead strandings ($n = 19$) showed a similar distribution, with 73.7% ($n = 14$) occurring between January and May (autumn: 53.3%, $n = 8$ and summer: 46.7% $n = 7$). April was the most prominent single-dead stranding month (28.6%, $n = 4$) followed by May, January and February (each 21.4%, $n = 3$). Mass strandings were more evenly distributed but autumn was again identified as the most prominent season (44.4%, $n = 4$), with March being the most common month for mass strandings (33.3%, $n = 3$), followed by September (22.2%, $n = 2$). Seasonality was also apparent in regards to stranding locations, with 75.0% ($n = 6$) of winter and spring strandings, regardless of group size, reported from the Hawke's Bay area (Figure 2.9). Likewise, 75.0% ($n = 3$) of winter and spring mass strandings occurred in the Hawke's Bay area. Conversely, the Chatham Islands with their prominent stranding history had no records of winter or spring strandings.

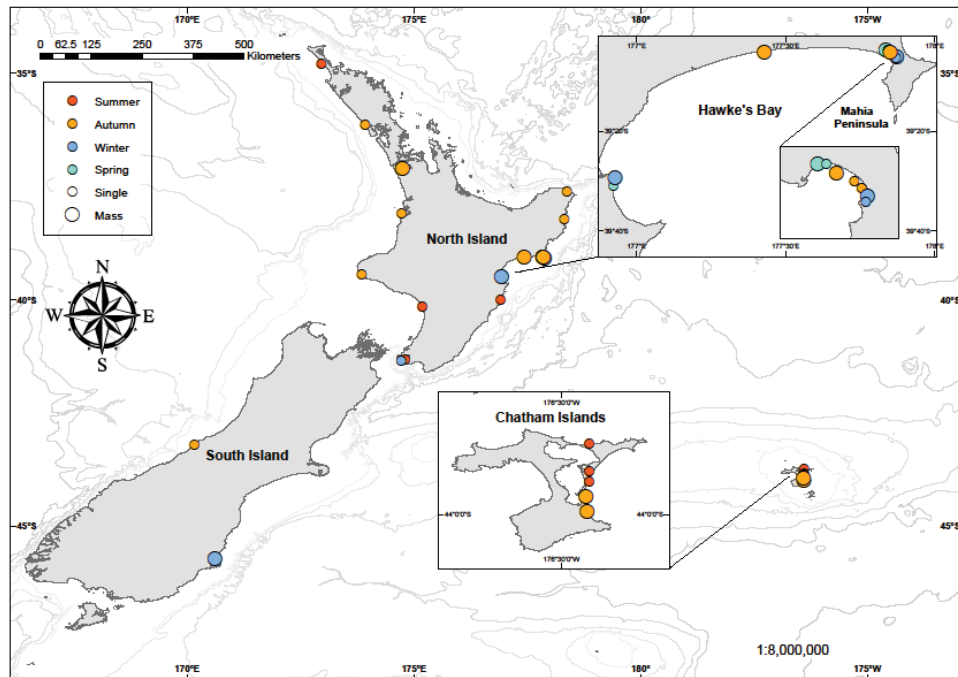


Figure 2.9. Seasonality of 28 false killer whale strandings between 1870 and 2005.

The occurrence of strandings in relation to SOI values was analysed, with records available for all strandings that occurred after 1876 ($n = 27$). As data distribution was abnormal (Kolmogorov-Smirnov $Z = 0.400$, $SD = 0.751$, $p = 0.010$), a discrete data test was applied in form of a Kruskal-Wallis test. Although trends were visible, these were not statistically significant ($Z = 43.08$, $d.f. = 62$, $p = 0.968$). Consequently, differences between SOI values in relation to mass or single strandings were not tested and trends rather than statistical significance are described.

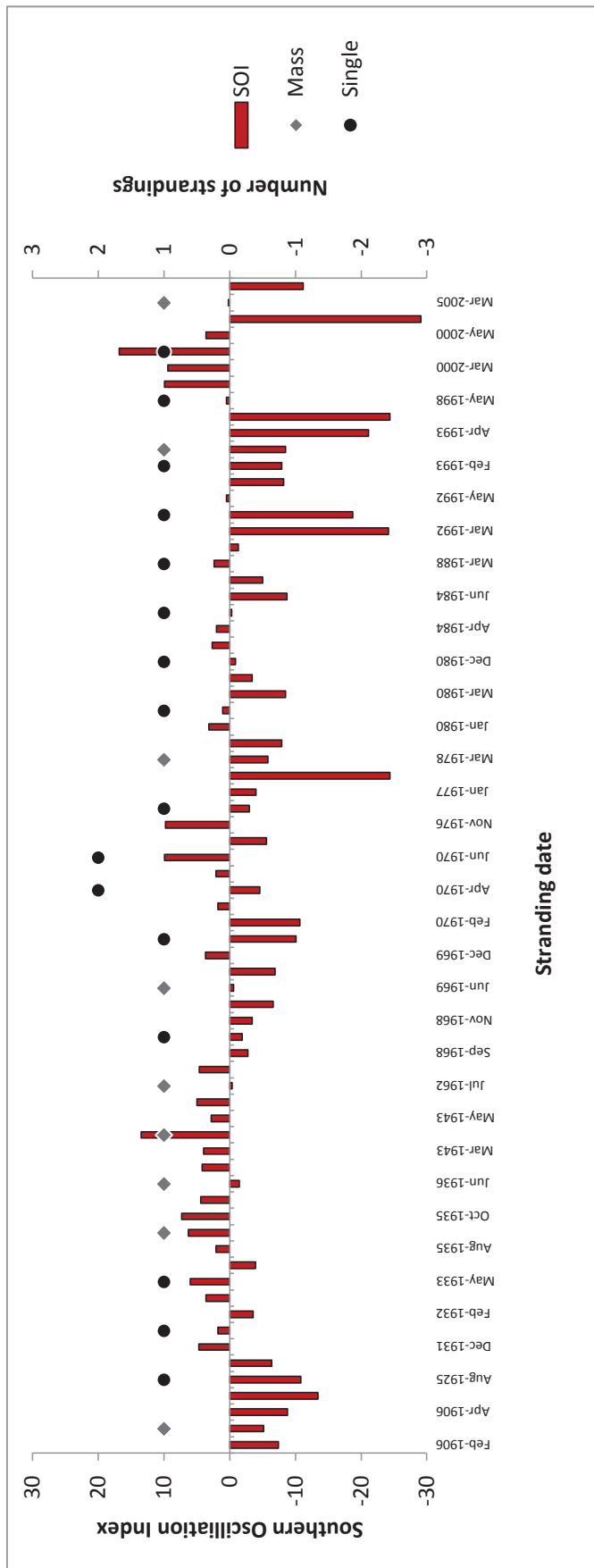


Figure 2.10. Documented mass (diamonds) and single (circles) strandings of false killer whales in New Zealand between 1906 and 2005 in relation to the SOI. Note: Displayed SOI ranges from one month prior to one month after each stranding event.

Strandings showed a slight positive trend in relation to negative SOI values (range: -18.7 – 16.8, \bar{x} = -0.6, median = -0.5, SD = 7.6, n = 27, Figure 2.10), with 18.5% of events (n = 5) recorded during El Niño episodes compared to 14.8% (n = 4) during La Niña events. Looking only at mass strandings (range: -8.5-6.3, \bar{x} = -1.7, median = -0.6, SD = 4.3, n = 9), 66.6 % of events (n = 6) occurred when SOI was < 0, with 88.8 % of events (n = 8) occurring when SOI was \leq 0.2 and 11.1 % (n = 1) during actual El Niño episodes compared to no sightings during La Niña events. Single strandings were more evenly distributed across a wider SOI range (range: -18.7-16.8, \bar{x} = -0.8, median = -0.6, SD = 8.4, n = 18), with 55.6% (n = 10) of events occurring when SOI was < 0 and a further 16.7% (n = 3) of single strandings each occurring during El Niño and La Niña conditions, respectively.

2.3.1.3 Group sizes and species composition at strandings

Group sizes ranged from 1 to ~300 (\bar{x} = 26.3, median = 1, SD = 72.09, n = 28, Figure 2.11). The relatively high overall mean (26.3 individuals) is skewed by the two largest events, with the nine documented mass strandings recorded to date, accounting for 97.3% of all individuals stranded. The low median (1) reflects that the majority of strandings (67.9%, n = 19) comprised of < 3 individuals, with 64.3% of all strandings (n = 18), comprising of singletons. The mean group size, taking into account the nine mass stranding events only, was 79.4 individuals (median = 29, SD = 113.4). Records show one joint mass stranding of false killer whales and pilot whales (presumably long-finned pilot whales), and another event involving pygmy sperm whales (*Kogia breviceps*), both occurring at Mahia Peninsula, Hawkes Bay.

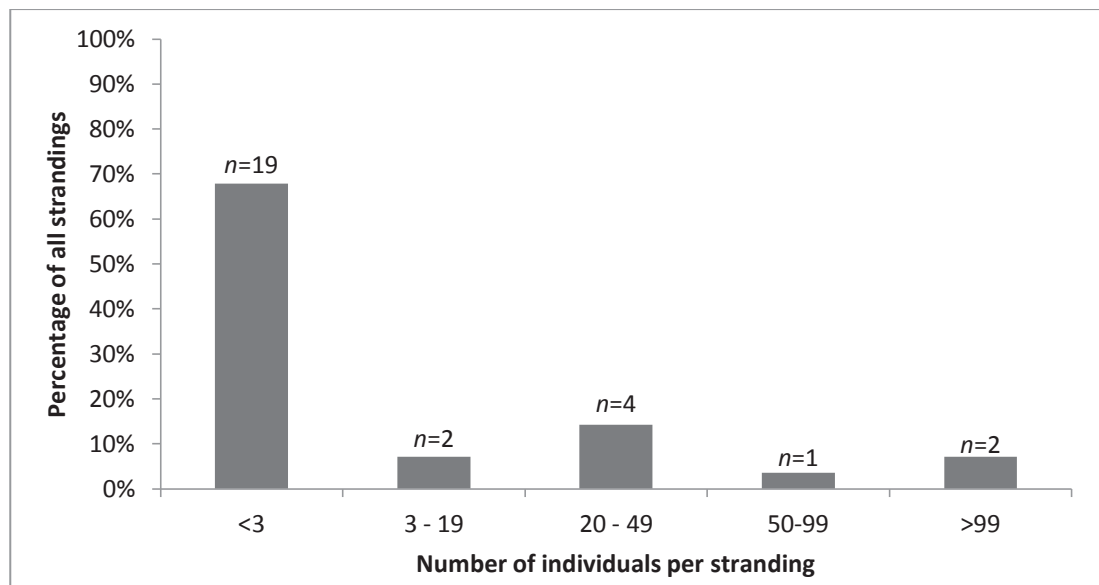


Figure 2.11. Number of individuals involved in the 28 false killer whales strandings in New Zealand since 1870.

2.3.2 At-sea observations

2.3.2.1 Tour operator poll

Of the 36 whale- and dolphin-watch operators and dedicated research platforms polled between June and July 2014 in regard to false killer whale sighting records, 66.7% ($n = 24$) responded, with 54.2% ($n = 13$) and 45.8% ($n = 11$) of respondents operating in the North and South Island, respectively. The vast majority of respondents (91.6%, $n = 22$) were tour operators, with 8.3% ($n = 2$, both North Island) from dedicated research platforms (Table 2.5). A total of 59 sighting records were reported. Of these 3.4% ($n = 2$) were classed *not verifiable* and omitted, resulting in 57 records from 21 operators and 2 research platforms.

Sighting reports indicated a strong trend for low latitude, with 100% ($n = 13$) of North Island platforms reporting verifiable sighting records of false killer whales, a sharp contrast to the complete lack of records from South Island waters (Figure 2.12).

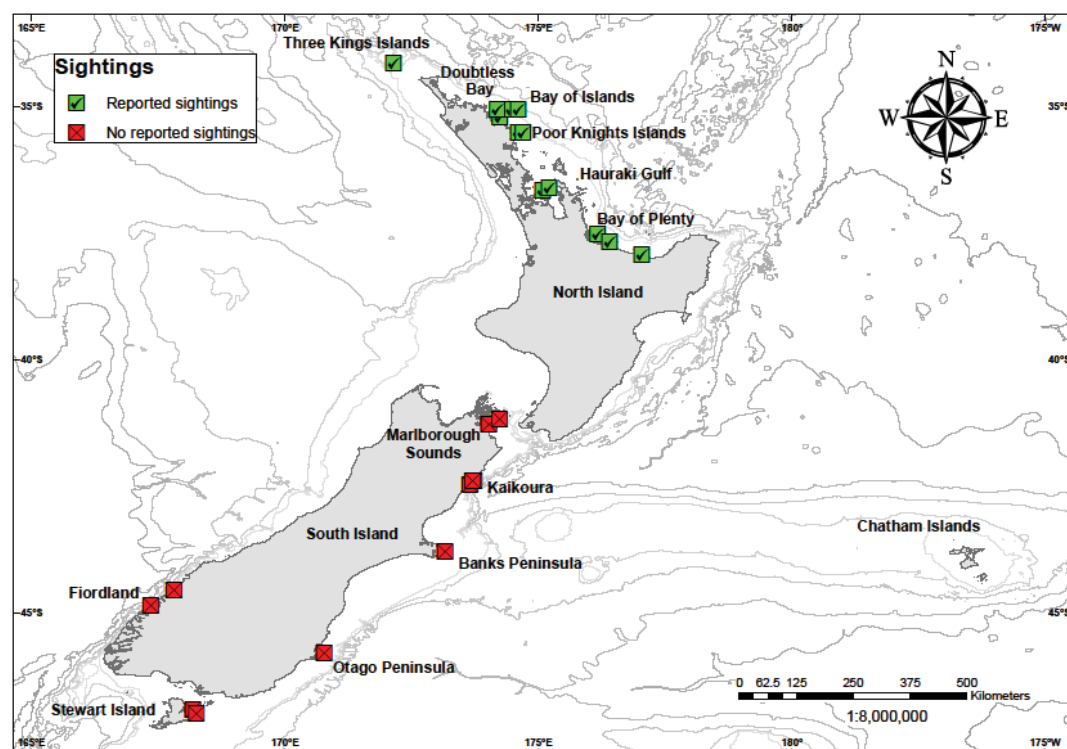


Figure 2.12. Locations of the 23 whale- and dolphin-watch operations and research platforms around New Zealand that provided false killer whale presence (green ticks) or absence (red crosses) reports.

Table 2.5. Presence/absence of false killer whale sighting records by whale-watch operators and dedicated research platforms (*) around New Zealand in their respective area of operation throughout the history of their operation. Poll conducted during June and July 2014. South Island and North Island waters were classed as *high latitude* and *low latitude*, respectively. Vessels that operated primarily in enclosed waters were considered *inshore*, while those in open waters were considered *offshore*.

North Island													
Name	Area of operation	Lat.	High/ low	Proximity	Oper. since	Season	Nr. of vessels	Fkw records	Avail. records	Records complete	Species verifiable	Date verifiable	Comments
Cascade Charters	Three Kings Is.	33°S	Low	Offshore	2007-2008	All year	1	Y	5	N	Y	Y	All records Apr-May
Dolphin Rendezvous	Doubtless Bay	34°S	Low	Offshore	2004-2009	Sep-May	1	Y	1	N	N	N	Unverifiable
Carno	Bay of Islands	35° S	Low	Inshore	1995	Sep-May	1	Y	3	Y	Y	Y	Duplicate records
Dolphin Discoveries	Bay of Islands	35°S	Low	Offshore	1992	All year	2	Y	7	N	Y	Y	Duplicate records
Ecocruz	Bay of Islands	35°S	Low	Offshore	2000	Oct-May	1	Y	4	Y	Y	Y	All records Dec-Apr
Great Sights	Bay of Islands	35°S	Low	Offshore	1994	All year	3	Y	29	Y	Y	Y	All records Dec-Apr
Dive Tutukaka	Poor Knights Is.	35°S	Low	Offshore	1999	All year	5	Y	1	N	Y	Y	Feb record
Ocean Blue	Poor Knights Is.	35°S	Low	Offshore	2006	All year	1	Y	1	N	Y	Y	Apr record
Dolphin Explorer	Hauraki Gulf	36°S	Low	Inshore	2002	All year	1	Y	1	Y	Y	Y	All records Jan-Mar
Te Epiwhani*	Hauraki Gulf	36°S	Low	Inshore	2010-2012	All year	1	Y	1	Y	Y	Y	Jan record
Aronui Moana	Bay of Plenty	37°S	Low	Offshore	2010-2012	All year	1	Y	2	Y	Y	Y	All records Jan-Feb
Dolphin Seafaris	Bay of Plenty	37°S	Low	Offshore	2006	Nov-May	1	Y	2	N	Y	Y	All records Jan-Feb
Pee Jay White Island T.	Bay of Plenty	37°S	Low	Offshore	1995	All year	3	Y	1	N	Y	Y	All records Jan-Mar
South Island													
Name	Area of operation	Lat.	High/ low	Proximity	Oper. since	Season	Nr. of vessels	Fkw records	Total records	Records complete	Species verifiable	Date verifiable	Comments
Dolphin Encounter	Kaikoura	42°S	High	Offshore	1995	All year	3	N	0	Y	N	N	Unverifiable
Dolphin Watch Tours	Marlborough S.	41°S	High	Inshore	2002	Oct-May	3	N	0	Y	N	N	
Golden Future	Marlborough S.	41°S	High	Inshore	1989	All year	1	N	0	Y	N	N	
Whale Watch Kaikoura	Kaikoura	42°S	High	Offshore	1987	All year	3-5	N	0	Y	N	N	
World of Whales	Kaikoura	42°S	High	Offshore	1990	All year	3	N	0	N	N	N	
Black Cat Cruises	Akaroa Harbour	43°S	High	Inshore	1988	All year	3	N	0	Y	N	N	
Real Journeys	Fiordland	44°S	High	Offshore	2002	All year	4-6	Y	1	N	N	N	
Monarch Wildlife Cr.	Otago Peninsula	45°S	High	Offshore	2002	All year	1	N	0	N	N	N	
Fiordland Expeditions	Fiordland	45°S	High	Offshore	2004	All year	2	N	0	Y	N	N	
Aihe Eco charters	Stewart Island	46°S	High	Offshore	1999-2008	All year	1	N	0	Y	N	N	
Rakiura Charters	Stewart Island	46°S	High	Offshore	2009	All year	2	N	0	Y	N	N	

The only reported sighting from South Island waters was a record from Fiordland. However, due to missing proof of species identification, the record was classed *not verifiable* and omitted. There were no other false killer whale records anywhere around the South Island, including off Kaikoura, where whale- and dolphin-watching in deep waters has been practiced on a daily basis since 1987. Based on proximity to shore, there was an even split of sightings, with 50% ($n = 3$) of inshore operators reporting sightings and lack thereof, respectively (Figure 2.13). Of the offshore operators, 52.9% ($n = 9$) reported sightings.

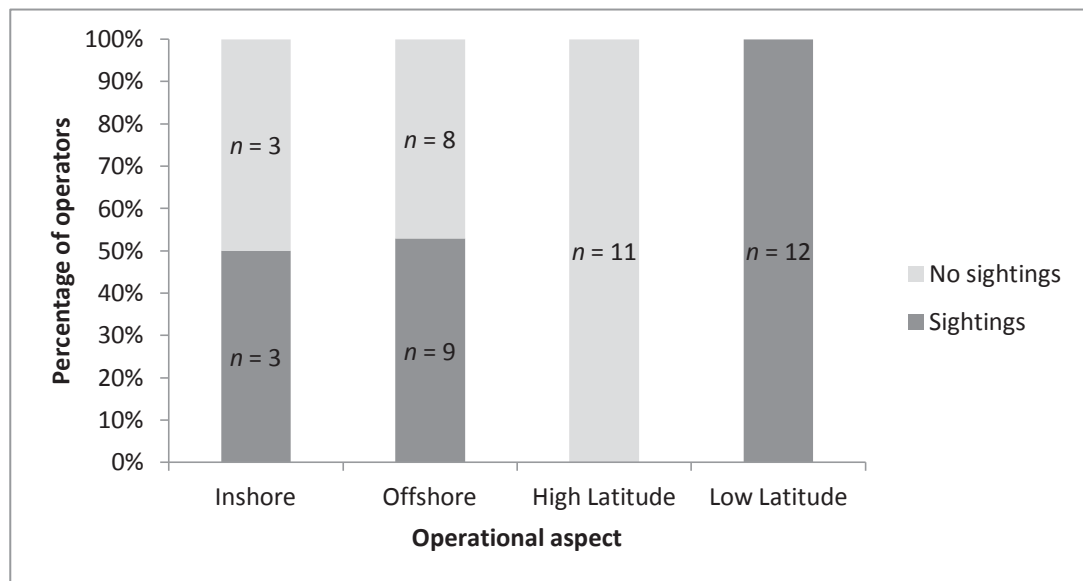


Figure 2.13. False killer whale sighting in relation to the operational aspect of 20 whale- and dolphin-watch operators from around New Zealand, showing a strong trend towards sightings at low latitudes.

2.3.2.2 Locations of reported sightings from tour and research vessels between 1995 and 2012

Fifty-seven sightings of false killer whales were recorded between 1995 and 2012 in waters off northeastern New Zealand. Of these, 17.5% ($n = 10$) were duplicates and omitted, resulting in 47 records (Table 2.6, Figure 2.14). There were no at-sea observations reported south of BOP despite the effort of 11 tour companies operating between the Marlborough Sounds and Stewart Island. The majority of observations (70.2%, $n = 33$) were made in BOI (2005, 2007, 2009 and 2010, with no false killer whales encountered in 2006, 2008, 2011, or 2012), with additional records from TKI (2008, 10.6%, $n = 5$), BOP (2009, 2012, 10.6%, $n = 5$), HG (2011, 4.2%, $n = 2$) and PKI (2010, 2011, 4.2%, $n = 2$). Encounter duration ranged from 10 minutes (min) to 225 min ($\bar{x} = 68.9$, $SD = 49.2$, $n = 47$).

During 91.5% of sightings ($n = 43$) false killer whales were observed in close proximity to bottlenose dolphins. Interspecific associations between false killer whales, bottlenose dolphins and other cetacean species are investigated in Chapter Four.

Table 2.6. Forty-seven documented false killer whale encounters off northeastern New Zealand between 1995 and 2007. Observations were made from tour boats (T) and/or research vessels (R). Area codes are Bay of Islands (BOI), Bay of Plenty (BOP), Hauraki Gulf (HG), Poor Knights Islands (PKI) and the Three Kings Islands (TKI). Codes for other species are *Globicephala melas* (Gm), *Orcinus orca* (Oo) and *Tursiops truncatus* (Tt).

Date	Area	Vessel type	Other species	Total group size	Group size Pc	Group size other species	Depth	SST	Encounter duration	Foraging observed
1995-Mar-15	BOI	T	Tt	150	n/a	n/a	100	20	120	N
1996-Feb-07	BOI	T	Tt	150	n/a	n/a	70	22	40	N
1998-Mar-16	BOI	T	Tt	400	n/a	n/a	101	23	105	Y
1998-Mar-21	BOI	T	Tt	100	50	50	120	20	70	N
1998-Apr-19	BOI	T	Tt	100	50	50	80	19	85	N
1999-Mar-10	BOI	T	Tt	200	n/a	n/a	48	23	75	Y
1999-Mar-16	BOI	T	-	20	20	-	40	23	35	Y
1999-Apr-04	BOI	T	Tt	150	n/a	n/a	112	20	50	Y
1999-Apr-05	BOI	T	Tt	150	n/a	n/a	60	20	75	N
1999-Apr-20	BOI	T	Tt	28	20	8	50	20	30	Y
2000-Jan-23	BOI	T	Tt	55	30	25	56	20	60	N
2000-Feb-07	BOI	T	Tt	150	n/a	n/a	60	22	80	N
2000-Mar-19	BOI	T	Tt	70	40	30	140	20	30	Y
2000-Mar-20	BOI	T	Tt	80	40	40	45	20	65	N
2000-Mar-21	BOI	T	Tt	100	50	50	90	22	45	N
2000-Mar-31	BOI	T	-	30	30	-	110	20	40	Y
2001-Mar-25	BOI	T	Tt	250	n/a	n/a	87	21	60	N
2001-Mar-27	BOI	T	Tt	200	n/a	n/a	85	21	65	N
2002-Apr-03	BOI	T	-	20	20	-	50	19	25	Y
2002-Apr-11	BOI	T	Tt	60	n/a	n/a	50	19	40	N
2002-Dec-20	BOI	T	Tt	130	80	50	76	18	50	N
2003-Feb-03	BOI	T	-	45	50	5	95	22	35	N
2003-Feb-07	BOI	T	Tt	80	30	50	60	22	55	Y
2003-Apr-09	BOI	T	Tt	200	100	100	95	19	45	N
2005-Feb-11	BOI	R	Tt	80	30	50	50	22	105	Y
2005-Mar-09	BOI	T	Tt	100	n/a	n/a	110	21	65	N
2005-Apr-23	BOI	T	Tt, Oo	80	30	Oo = 3, Tt = 30	60	20	90	N
2007-Jan-11	BOI	R	Tt, Gm	100	50	Tt = 50, Gm = 20	75	22	210	N
2007-Apr-03	BOI	R	Tt	80	30	50	110	20	165	Y
2007-Dec-28	BOI	R	Tt	150	50	100	120	21	225	Y
2008-Apr-25	TKI	T	Tt, Gm	90	30	Tt = 40, Gm = 20	350	19	30	N
2008-Apr-27	TKI	T	Tt	180	100	50	340	19	15	N
2008-May-06	TKI	T	Tt	150	100	50	330	19	10	N
2008-May-16	TKI	T	Tt	100	40	60	350	19	20	N
2008-May-17	TKI	T	Tt	100	40	60	330	19	100	Y
2009-Feb-25	BOP	T	Tt	70	35	20	35	20	45	Y
2009-Dec-21	BOI	R	Tt	110	50	60	115	20	180	Y
2010-Mar-20	BOI	T	Tt	80	30	50	100	21	20	N
2010-Mar-25	BOI	R	Tt, Oo	75	60	Tt = 5, Oo = 8	25	21	120	N
2010-Apr-20	PKI	T	Tt	60	30	30	60	20	20	N
2011-Jan-20	HG	T	Tt	300	150	150	45	22	60	Y
2011-Jan-25	HG	R	Tt	90	30	60	46	22	118	Y
2011-Feb-12	PKI	T	Tt	80	30	50	100	23	30	N
2011-Mar-03	BOP	T	Tt	120	40	80	150	19	30	N
2012-Jan-18	BOP	R	Tt	170	20	200	80	20	120	Y
2012-Feb-09	BOP	R	Tt	120	20	150	35	20	100	Y
2012-Mar-10	BOP	T	Tt	90	40	50	80	19	40	Y

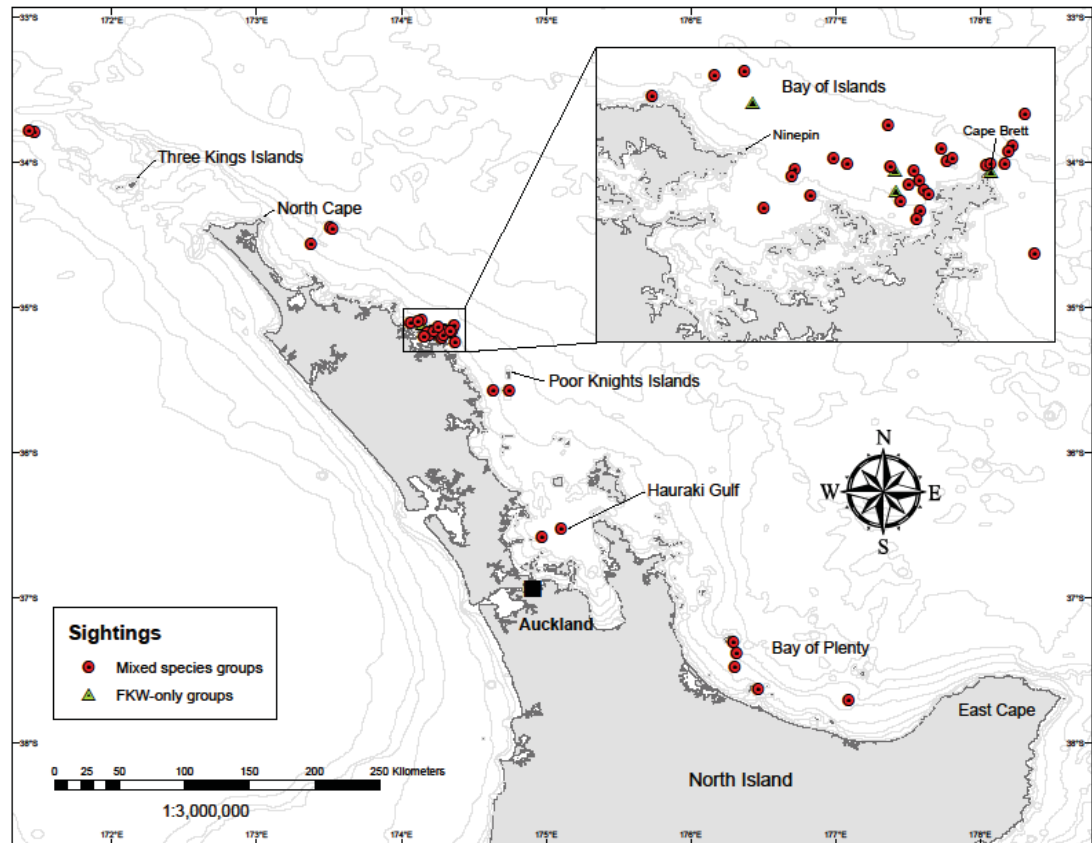


Figure 2.14. Locations of opportunistic false killer whale sightings between 1995 and 2012 off northeastern New Zealand. Red circles indicate mixed species groups of false killer whales and bottlenose dolphins. Green triangles indicate groups of false killer whales only.

2.3.2.3 Sea surface temperature, depth and distance from shore

False killer whales were encountered in SST ranging between 18 and 23° C (\bar{x} = 20.5° C, SD = 1.3, n = 47). However, SST had no significant effect on the number of encounters (Kruskal-Wallis, Z = 4.86, d.f. = 5, p = 0.434). Examining only BOI records, the area with the largest sample size, SST also ranged between 18 and 23° C (\bar{x} = 21° C, SD = 3.7, n = 29), with 86.2% (n = 25) of observations made when SST was $\geq 20^\circ$ C (Figure 2.15).

Bottom depth for the sightings ranged from 25 to 350 m (\bar{x} = 105.3 m, SD = 86.7, n = 47) with 63.8% of encounters (n = 30) occurring in waters < 100 m deep. Distance from shore ranged from 0.2 to 67.4 km (\bar{x} = 9.2 km, SD = 14.8, n = 47).

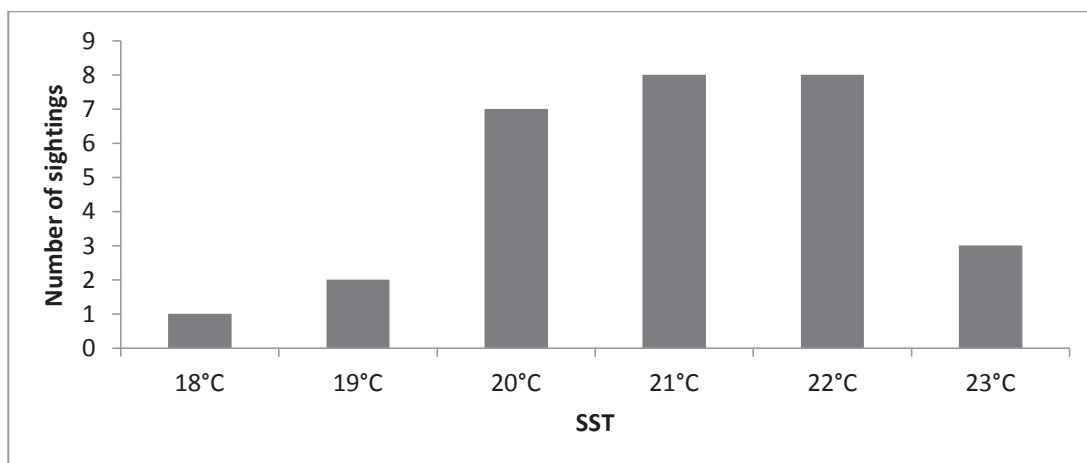


Figure 2.15. False killer whale sightings in relation to SST, recorded in the Bay of Islands, New Zealand, between January 1995 and June 2007.

2.3.2.4 Seasonality

False killer whales were rarely encountered. Records collected aboard *Tutunui* in BOI state 29 sightings during 6,108 trips on 4,082 discrete days during the vessel's year round operation between 1995 and 2007. The overall TER was 0.47 encounters per 100 trips. No significant differences were found between months ($Z = 4.73$, d.f. = 11, $p = 0.943$) and years ($Z = 3.66$, d.f. = 12, $p = 0.989$) but were identified between warm and cold seasons ($Z = 7.64$, d.f. = 1, $p = 0.006$). Sightings in BOI only occurred during the austral summer (TER = 0.37, $n = 8$) and autumn (TER = 1.33, $n = 21$) with TER highest in March (TER = 2.04, $n = 12$) and April (TER = 1.98 $n = 10$, Figure 2.16).

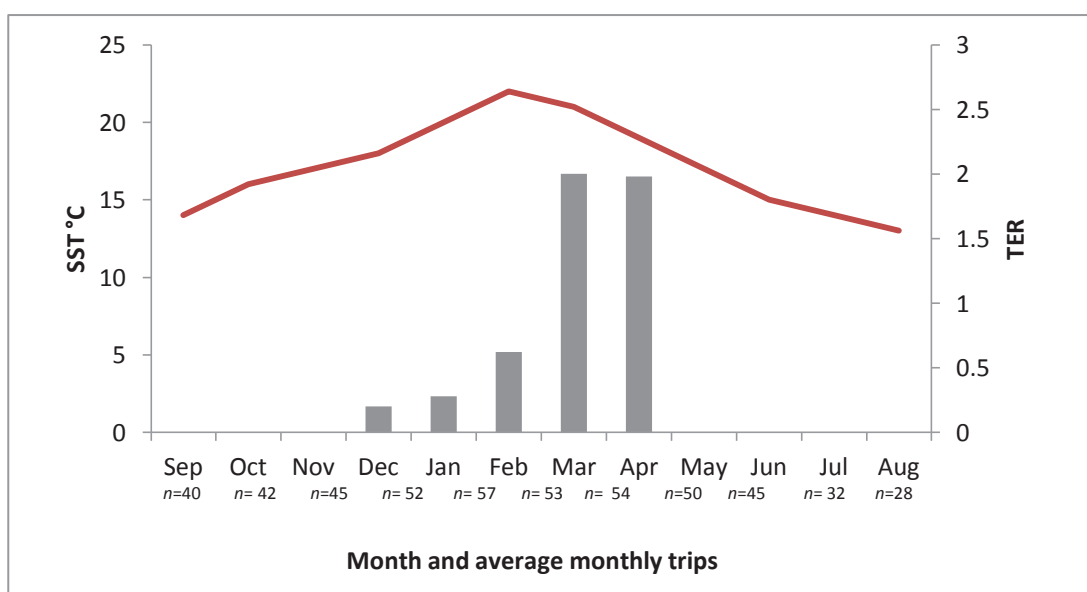


Figure 2.16. Mean monthly sea surface temperature (SST) and trip encounter rate (TER) of false killer whales in the Bay of Islands, New Zealand, January 1995- June 2007.

Seasonality for sightings in the other locations in the study area was as follows: TKI, autumn, $n = 5$, PKI, summer, $n = 2$, HG, summer, $n = 2$, BOP, summer, $n = 3$ and autumn, $n = 2$ (Figure 2.17).

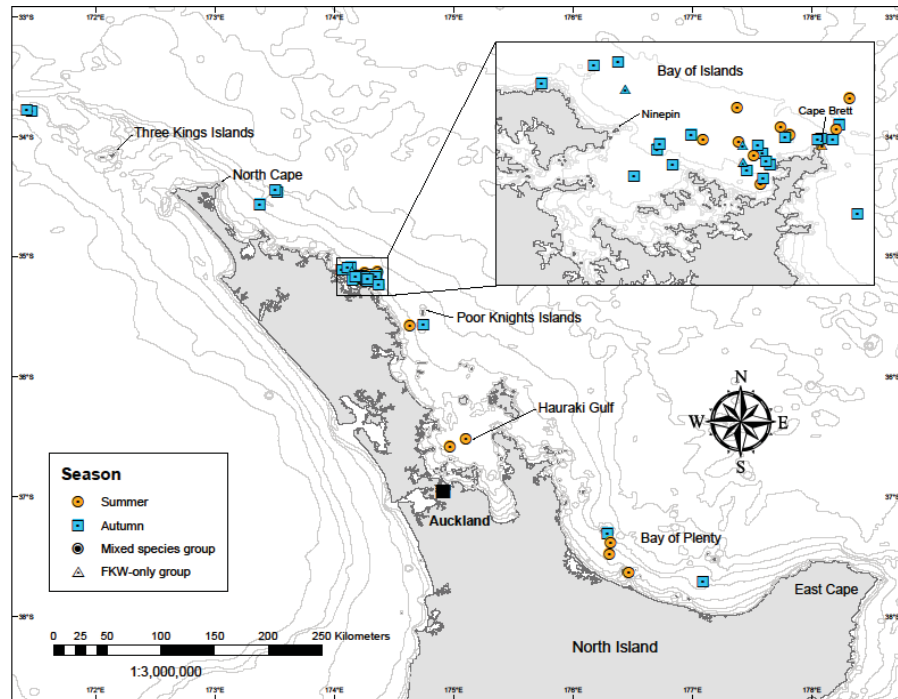


Figure 2.17. False killer whale sightings off northeastern New Zealand between 1995 and 2012 by season.

BOI Sightings in relation to SOI (range: -29.1-18.5, $\bar{x} = -0.8$, median, 1.1, SD = 13.8, $n = 29$, Figure 2.18) showed that the majority of false killer whales were observed during positive SOI values (55.1%, $n = 16$), with 34.4% ($n = 10$) observed during La Niña and 20.1% ($n = 6$) during El Niño events. However, visible trends were not statistically significant (Kruskal-Wallis: $Z = 46.85$, d.f. = 140, $p = 1.000$).

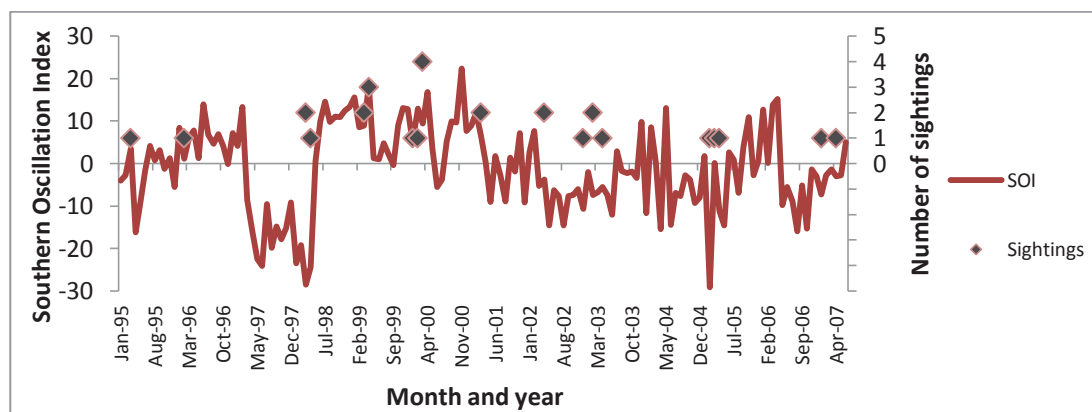


Figure 2.18. False killer whale sightings in the Bay of Islands, New Zealand between January 1995 and June 2007 in relation to the southern oscillation index (SOI).

2.3.2.5 Group sizes

Mixed-species group sizes ranged from 28 to *ca.* 400 (\bar{x} = 120.4, SD = 64.6, n = 43) (Figures 2.19 & 2.20). Within those mixed-species groups, species specific group size estimates were recorded during 76.6% of encounters (n = 36), with group size for false killer whales ranging from 20 to *ca.* 150 individuals (\bar{x} = 46.7, SD = 28.5). In comparison, group size estimates for false killer whales observed in single-species groups ranged from 20 to 50 individuals (\bar{x} = 35.0, SD = 12.9, n = 4).



Figure 2.19. A mixed-species group of at least 36 false killer whales with some bottlenose dolphins at the periphery in Bay of Islands, New Zealand, December 2007. Photo © Daniel Panek.

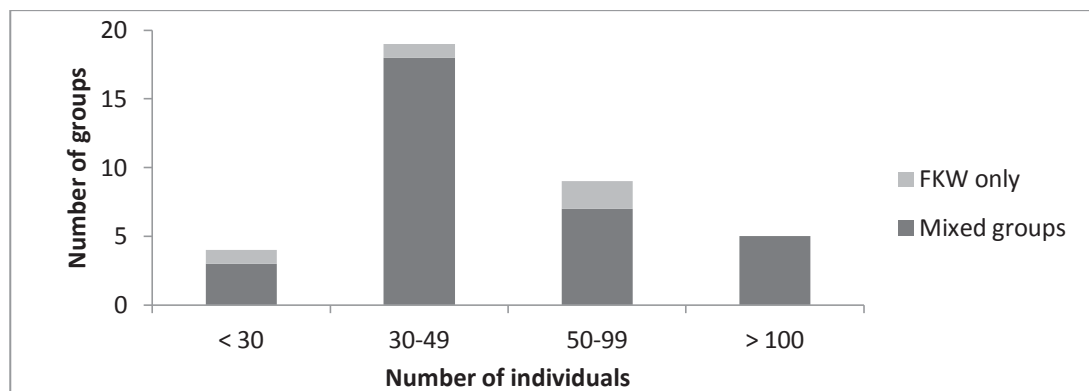


Figure 2.20. Group sizes of false killer whales (FKW) encountered off northeastern New Zealand between 1995 and 2012 in mixed-species groups with bottlenose dolphins (dark grey) and in false killer whale-only groups (light grey).

2.3.2.6 Foraging observations

Foraging was observed during 42.6 % ($n = 20$) of all 47 encounters reported here and during 75.0% ($n = 3$) of the single-species observations of false killer whales ($n = 4$). Within the mixed-species observations ($n = 43$), foraging was observed during 39.5% ($n = 17$) of encounters. Mixed-species groups involved in foraging were generally larger ($\bar{x} = 135.8$, $SD = 92.9$, $n = 17$) than those groups where foraging was not observed ($\bar{x} = 112.5$, $SD = 57.1$, $n = 26$) (Figure 2.21).

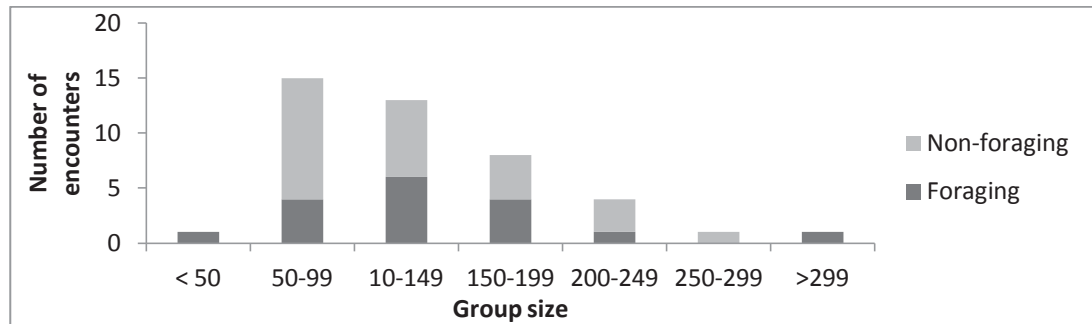


Figure 2.21. Group sizes of 43 mixed-species groups of false killer whales and bottlenose dolphins encountered off northeastern New Zealand between 1995 and 2012 in relation to the absence (light grey) and presence (dark grey) of observed foraging behaviour.

However, encounters where foraging was observed were generally longer ($\bar{x} = 82.5$, $SD = 55.4$, $n = 20$) than those where it was not detected ($\bar{x} = 58.9$, $SD = 42.3$, $n = 27$, Table 2.8, Figure 2.22). A negative binomial regression found no evidence that encounter duration ($p = 0.994$) and foraging behaviour ($p = 0.420$) had any significant effect on group size estimates (Table 2.7). Evaluation of residuals found the assumption for homogeneous variance and normal distribution of residuals to be met and p-value considered reliable.

Table 2.7. Negative binomial regression investigating potential predictors of false killer whale group size estimates off northeastern New Zealand between 1995 and 2012.

Parameter	Estimate	Std. Error	Z-value	Pr (> z)
<i>Intercept</i>	4.782000	0.135800	35.212	< 0.001
<i>Encounter Duration</i>	-0.000012	0.001593	-0.007	0.994
<i>Foraging Behaviour</i>	0.113030	0.161600	0.806	0.420

Null deviance: 45.216 on 42 degrees of freedom

Residual deviance: 44.496 on 40 degrees of freedom

Model: $NB_{\text{Group Size Estimate}} = \alpha + \beta_{\text{Encounter Duration}} + \beta_{\text{Foraging Behaviour}} + \varepsilon$

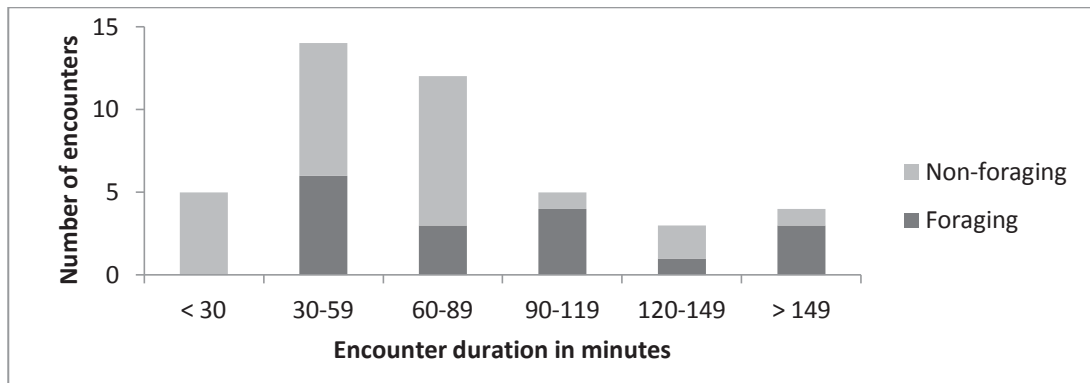


Figure 2.22. The absence (light grey) and presence (dark grey) of foraging behaviour in relation to the duration of the observation of mixed-species groups of false killer whales and bottlenose dolphins encountered off northeastern New Zealand between 1995 and 2012.

False killer whale prey species were confirmed to be kahawai (*Arripis trutta*, BOI $n = 3$), yellowtail kingfish (*Seriola lalandi*, HG $n = 2$, BOP $n = 1$) and hapuku (*Polyprion oxygeneios*, $n = 1$ TKI, Figure 2.23).



Figure 2.23. A false killer whale feeding on hapuku (*Polyprion oxygeneios*) on the King Bank, ca. 150 km northwest of New Zealand. Photo © David Donnelly, 2008.

The joint herding of kahawai by false killer whales and bottlenose dolphins was observed during 3 encounters (BOI 2007, $n = 1$ and HG 2011, $n = 2$, Figure 2.24). During the two HG observations, members of both species were observed to encircle a large school of kahawai, using a carouselling technique (Bel'kovich *et al.* 1991). Additionally, the emission of bubble bursts was observed along the periphery of the school. During this period, the fish school was driven closer to the observation vessel, with the hull eventually acting as a physical

barrier that restricted the schools horizontal movements. The removal of the head, entrails and subsequent food sharing of kingfish, characterised by one individual holding the fish whilst others dislodged chunks, was observed during a further two encounters (BOI).



Figure 2.24. A false killer whale and a bottlenose dolphin encircling a shoal of kahawai using a carouseling technique (A). Bubble bursts used to herd a shoal of kahawai into a tight “bait ball” (B). The shoal of kahawai being herded against the hull of the observation vessel (C). A false killer whale holding a kahawai in its jaws (D). Hauraki Gulf, New Zealand, January 2011. Photo © Sarah Gardner

2.4 Discussion

2.4.1 Occurrence in New Zealand waters

2.4.1.1 Strandings

The infrequency of strandings of a species known for its prominent stranding tendency in other regions (Odell and McClune 1999), together with the broad spatial spread of those events, suggest a wide yet scarce false killer whale distribution in New Zealand waters. Only 28 stranding events have been documented to date, compared, for example, to 454 common dolphin, 391 pygmy sperm whale, and 370 pilot whale strandings recorded in the same period (1840 - 2014).

Spatial distribution of stranding locations indicates a notably higher concentration on the North Island's east coast and the Chatham Islands. Assessment of mass strandings alone further supports these results, with only two events documented in different areas (upper North Island west coast and lower South Island east coast, respectively). However, under-reporting from the North and South Island's west-coasts, due to remoteness and low population density has to be considered. The two prime false killer whale stranding locations, the Mahia Peninsula in Hawkes Bay and the Chatham Islands, are also considered stranding 'hot spots' for a variety of other cetacean species, in particular pilot whales (Brabyn 1991). However, no false killer whale strandings have been documented from the other two stranding 'hot spots', Whangarei and Farewell Spit.

While the lack of reports from the South Island's Farewell Spit area may be indicative of occurrence that is centred further north, the lack of strandings from northeastern New Zealand's Whangarei area is harder to explain. Interestingly, Whangarei lies in the centre of the area along the northeastern coast of New Zealand, where all at-sea observations collated in this study were documented. This may simply be due to the fact that false killer whale strandings appear to be rare events in New Zealand.

While cetacean strandings may provide an indication of species abundance and richness (Pyenson 2010, Thompson *et al.* 2013), it has also been suggested that stranding locations may not reliably reflect a species' natural distribution, as sickness may cause individuals to stray outside of their normal ranges and strand (Leatherwood *et al.* 1989). However, Maldini *et al.* (2005) suggest that strandings constitute reasonable indicators of frequency of occurrence based on comparisons of stranding records with sighting surveys in the corresponding areas, and Thompson *et al.* (2013) argue that strandings may be more accurate at documenting rare species that are often overlooked during sighting surveys. Brabyn (1991) further argues that single-dead strandings may serve as an approximate indicator for a species natural distribution, given that they invariably involve individuals that have died at sea and washed ashore. However, this is debatable as it may simply be the result of an individual moving beyond its natural home range due to sickness.

The stranding data assessed herein suggest that false killer whales are uncommon in waters off the South Island, which reflects the preferred warm water distribution patterns reported from other regions (Baird 2008).

2.4.1.2 At-sea observations

A similar apparent trend was also observed in the reported at-sea observations. Sighting information from whale-watch operators around the country showed that latitude was a clear

determinant of false killer whale sightings, with no reported records from South Island waters. This may be a further indicator that the species' occurrence in New Zealand waters is likely centred further north.

It is also unclear how common false killer whales are south of East Cape. Unfortunately, no sighting reports exist from the Hawke's Bay region, due to the lack of suitable platforms of opportunity. However, the prominent stranding history in the area suggests regular occurrence in those waters. Furthermore, the area's geographic location within the reaches of the warm East Cape Current is consistent with false killer whales' preferred SST range.

2.4.1.3 Occurrence off northeastern New Zealand

The at-sea observations from northeastern New Zealand waters reveal false killer whale occurrence well within the continental shelf and < 100 m in bottom depth. Despite their reported pelagic distribution (Wade and Gerrodette 1993, Baird 2008), false killer whales are known to approach close to shore at oceanic islands (Baird *et al.* 2010), with forays into shallow continental shelf waters also reported (Acevedo-Gutiérrez *et al.* 1997, Palmer *et al.* 2009, Weir *et al.* 2013). While the latter are thought to be uncommon (Baird *et al.* 2008), results presented in this chapter suggest that they may be more frequent than previously assumed. Although the sightings in relatively shallow coastal waters are purely a function of effort and not representative of actual habitat use, they nevertheless show that this species ventures into such areas with some regularity. The extremely low encounter rate in the study area also indicates that the species' distribution in New Zealand waters is likely centered further offshore, which would be consistent with the distribution generally reported for false killer whales (Baird 2008). Consequently, false killer whales encountered in the study area are likely to frequent deep oceanic, as well as the shallow continental shelf waters in which most of the observations reported here were made.

2.4.2 Seasonality

While stranding reports exist from all months except November, suggesting year-round occurrence in New Zealand waters, seasonality was apparent, with the majority of strandings occurring between December and June. At-sea observations showed a similar trend. While seasonality could only be assessed from records in BOI, all encounters from the different locations within the study area fall within the same period (December – May), with the vast majority of encounters occurring at SST of $\geq 20^{\circ}\text{C}$, indicating that occurrence in nearshore waters is likely seasonal.

Similarly, at-sea observations of false killer whales in the study area coincide with the seasonal flow of the EAUC, a warm current, which begins its shoreward progression towards northeastern New Zealand in December and completes its annual cycle by May (Zeldis *et al.* 2004). SST within the EAUC is *ca.* 2° C warmer than on the continental shelf (Sharples 1997). This current is associated with the arrival of warm water species such as various *Thunnus* and *Istiophoridae* species (Francis *et al.* 1999), the latter two being known prey items for false killer whales (Baird 2008). In Japanese waters, seasonal changes in occurrence have been reported (Kasuya 1971) and movements of false killer whales have been linked with warm water masses and migrations of prey (Tomilin 1957, Kasuya 1971, Leatherwood *et al.* 1989). Results presented herein support the hypothesis that the seasonal occurrence of false killer whales in waters of northeastern New Zealand may be the result of the species following the shoreward flow of the EAUC, presumably in pursuit of prey.

The fact that all early and late season reports were from the northernmost parts of the study area may be indicative of the species movements during colder months but could be merely effort related. However, movements towards the north would be consistent with the species' general preference for warmer waters, and further supports the hypothesis that false killer whales in the study area may be associated with seasonally shifting warm water masses. While there was a slight positive trend of sightings during La Niña episodes, compared to a slight negative trend of strandings in El Niño conditions, these may simply be an artifact of the small sample size, which precluded meaningful analysis of the effect of long-term weather patterns on false killer whale occurrence in the study area.

2.4.3. Group size

Group size estimates of false killer whales observed at sea appear to be larger than those reported from most other areas (refer to Chapter One), except for Japanese waters (Kasuya 1986). While there was little discrepancy in average group sizes of at-sea encounters and strandings in New Zealand, some of the stranded groups were significantly larger than those reported from at-sea observations. A similar scenario has been reported from South Africa (Best 2007, Ferreira 2008, Kirkman *et al.* 2010). It has been suggested that the larger group sizes recorded at strandings may be the result of the formation of temporary feeding aggregations in nearshore waters due to abundant prey availability (Bradshaw *et al.* 2006, Ferreira 2008).

A causal link between large group size and the occurrence of mass stranding events has also been suggested, albeit indirectly, in pilot whales. Oremus *et al.* (2013) hypothesised that large, temporary feeding or mating aggregations of unrelated social groups may facilitate

competition and/or aggressive behaviour between individuals of different groups, causing disruption of kinship bonds within their own social units, thus perhaps influencing the occurrence of large mass strandings in the species. The suggestion is based on their study of kinship in stranded long-finned pilot whales, which confirmed the presence of multiple matriline, indicating that mass strandings may comprise more than one social unit (Oremus *et al.* 2013).

Similar factors may explain the larger group sizes documented for mass stranded false killer whales. However, false killer whales can disperse for more < 20 km and still be considered part of the same group (Baird *et al.* 2010). Such group dynamics make accurate size estimation difficult and increase the chances of negative bias. Additionally, observer error has to be considered when estimating large group sizes containing more than one species. A comparison of group size estimates of odontocetes by experienced boat based observers and counts from aerial photographs of the same groups taken from a helicopter, showed that at-sea observers tend to underestimate group size by *ca.* 26% (Gerrodette *et al.* 2002). Consequently, the large group sizes reported herein may simply be a result of observer bias and more records from experienced observers are needed to ascertain their validity. However, it is worth noting that the largest group sizes reported from other regions, representative of at-sea observations, have been reported from Japanese drive hunts (up to 201 individuals), thus presenting accurate counts rather than estimates (Kasuya 1986).

Gygax (2002) suggests a causal link between SST and group size in delphinids, with large groups more likely to occur at lower SST. More abundant and/or less evenly distributed food sources in cooler waters have been suggested as possible drivers behind the larger groups observed. As New Zealand waters appear to be at the limit of false killer whales' preferred temperature range, this may be a contributing factor behind the larger groups observed. False killer whale group sizes in New Zealand waters appear to be comparatively large, and may yet be underestimated. Likewise, sizes of stranded groups may be negatively biased due to the possibility of omitting individuals that were successfully refloated or managed to refloat themselves, especially at the early stages of a mass stranding. The size estimates of stranded groups should therefore be considered as the minimum size. More data are required to ascertain the reason behind the larger group sizes recorded during some stranding events.

2.4.4 Foraging observations

Foraging observations in the study area documented some previously unrecorded prey species and foraging techniques (see also Zaeschmar *et al.* 2013, Appendix C). Predation on kahawai, a schooling coastal species endemic to areas within temperate Australasian waters

(Paulin 1993), has so far not been documented from other regions for false killer whales. While predation on a coastal fish species may simply constitute opportunistic feeding, it may also suggest that foraging in nearshore or inshore waters may occur more frequently than is currently being reported. Additionally, false killer whales were also observed to be feeding on kingfish in inshore waters and on hapuku in offshore waters. Predation on kingfish, a large pelagic fish inhabiting coastal and oceanic waters, and hapuku, a large demersal fish, inhabiting deep waters off the continental slope (Beentjes and Francis 1999), is more consistent with the feeding ecology reported for the species from other regions (Odell and McClune 1999). It is therefore possible that the seasonal shoreward flooding of warm currents allows false killer whales access to a food source that may otherwise be outside of their preferred temperature range.

The larger group sizes recorded during foraging observations may be the result of widely dispersed groups contracting when abundant prey is encountered. Indeed, satellite tagging in Hawaiian waters revealed that groups may disperse over more than 20 km (Baird *et al.* 2010). Food sharing and the discarding of entrails, tails and gills, as observed in the study area, has also been described from other regions (*e.g.* Shallenberger 1981, Baird *et al.* 2008). While the use of bubbles during prey capture has been widely described for mysticete species, in particular the humpback whale (*Megaptera novaeangliae*) (*e.g.* Sharpe and Dill 1997), it has not so far been reported for false killer whales. However, false killer whales have been observed to use bubbles underneath an observation vessel in Hawaiian waters where mahi mahi (*Coryphaena hippurus*) were known to be present. In this case, it was assumed that the bubbles were used to dislodge fish attempting to evade predation (Robin Baird, pers. comm.⁴). Such foraging behaviour does not appear to be common among delphinids in general, although at least one record exists for the bottlenose dolphin (Fertl and Wilson 1997) and it has also been observed in several other cetacean species such as Atlantic spotted dolphin (*Stenella frontalis*) (Fertl and Würsig 1995), short-beaked common dolphin (*Delphinus delphis*, Neumann and Orams 2003), dusky dolphin (Trudelle 2010) and killer whale (Similä and Ugarte 1993). In contrast to the frequent observations of joint foraging by false killer whales and bottlenose dolphins in the study area, only very few records of false killer whales foraging in mixed-species associations with other cetaceans exist in the literature (*e.g.* Tsutsumi *et al.* 1961).

⁴ Robin W. Baird, Cascadia Research Collective, 2181/2 West 4th Avenue, Olympia, WA 98501, June 2011.

2.4.5 Study limitations

The stranding data analysed herein present a number limitations. Under-reporting of strandings due to possible species misidentification, especially confusion with pilot whales, has to be considered. The February 2005 stranding of 52 false killer whales at Long Beach on the Chatham Islands is a case in point; originally misidentified as pilot whales, correct identification was only made opportunistically *ca.* 2 months after the event during the examination of the remaining carcasses by another researcher (Clinton Duffy pers.comm⁵, Figure 2.25).



Figure 2.25. A stranded false killer whale, part of a group of 52 individuals at Long Beach, Chatham Islands in February 2005 (DOC reference 13/02/2005PSEL). Initially misidentified as pilot whales, correct identification was made opportunistically in April 2005, highlighting the risk of misidentification and subsequent under-reporting of the species both at strandings and at sea. Photo © Clinton Duffy.

Furthermore, there are some discrepancies between the stranding records assessed in the present study and those analysed by previous scholars. In Brabyn's (1991) analysis of the

⁵ Clinton Duffy, Marine Ecosystems Team, Science & Capability Group, Department of Conservation, Tāmaki Makaurau Auckland Office, PO Box 68908, Newton, Auckland 1145, May 2014.

New Zealand Stranding Record, five strandings events, involving 413 individuals (not included in the analysis here), were simply listed as ‘Blackfish: pilot/false killer’. The 2014 stranding tally obtained from DOC for the present analysis no longer included a ‘blackfish’ category and it could not be verified if these records were added to the ‘unidentified pilot whale’-, ‘unidentified toothed whale’-, one of the other two pilot whale categories (long- or short-finned) or omitted altogether. As the total number of stranded false killer whales in the 2014 data set is actually smaller than that given in Brabyn’s 1991 report (735 vs. 760), it can be assumed that none of the individuals from the ‘blackfish’ category were added to the false killer whale total.

The discrepancy between the 1991 and 2014 false killer whale tallies is also puzzling. The 1991 census puts the number of individuals at 760 in 20 discrete events, while the 2014 data shows 676 individuals in 22 discrete events for the same period (1840 – 1991). According to the 2014 data set, there have been six stranding events involving 59 individuals since 1991. It stands to reason that misidentification and/or reporting continues to be an issue, affecting accurate stranding figures and is likely not restricted to false killer whales. It is, therefore, difficult to ascertain if this will result in over- or under-reporting of false killer whale strandings. However, as false killer whales are an uncommon and little known species in New Zealand, it can be assumed that confusion with the considerably more prevalent pilot whales is the more likely possibility. The actual number of false killer whale stranding events and individuals involved is therefore likely higher than stated.

It has to be emphasised that the majority of consulted false killer whale stranding records are not supported by any form of proof of species (*e.g.* photographs, biopsy samples, bones *etc.*) and that species designation herein relies largely on the unspecified verification efforts carried out by previous scholars (Baker 1981, Brabyn 1991). Consequently, the fine scale results, in particular, have to be viewed with caution and the focus should lie on the broad scale findings, which indicate that false killer whale strandings are rare events, occurring mostly on North Island shores. As such, results presented do support the hypothesis that false killer whale occurrence in New Zealand waters is largely restricted to waters north of the SC. The two documented stranding records from South Island shores may therefore be best considered to be extralimital. These findings highlight the great need for accurate species identification at strandings and an improvement of the subsequent archiving of stranding data. Genetic sampling of stranded individuals is highly recommended, together with photographs identifying both species and individuals.

Likewise, the limitations presented by the opportunistic nature of the majority of sighting records analysed here have to be taken into account. As such, the apparent predominant

occurrence in northern New Zealand waters may simply be due to insufficient effort in other areas. Similarly, the apparent seasonality may be the result of inconsistent sampling effort, restricted to a small study area (BOI) that is not representative of the species' known preferred deep water habitat. Size estimates of groups encountered at sea may be affected by factors such as encounter duration, sea state, arc of vision, observer height and observer skills. Likewise, quantification of reported foraging observations has to be viewed with caution. The small sample size combined with the heterogeneity of the data collection meant that the stratification of results would further compromise its data quality. Consequently, fine scale analysis has been largely omitted in favour of focusing on the more robust broad scale trends achieved by the large temporal and spatial aspect of the study.

2.4.6 Conclusion

Stranding and at-sea observations presented herein indicate that false killer whales are rarely encountered in New Zealand waters. Spatial and seasonal trends further suggest that false killer whale occurrence in New Zealand waters may be influenced by ocean currents and fronts. Occurrence in the region appears most frequent in the EAUC influenced waters off the North Island, in particular the east coast north of the SC. These findings correspond with the species' reported preference of the warmer, lower latitude waters of its range. Furthermore, occurrence in nearshore waters of the BOI show a strong seasonal aspect, possibly linked to the seasonal shoreward flooding of the warm EAUC. Seasonal occurrence in nearshore waters is further supported by sightings in the other study locations off northeastern New Zealand and, albeit to a somewhat lesser extent, by New Zealand-wide stranding events. Average group sizes appear larger than those documented in other regions but still lie within the range reported for the species in general. Foraging observations show that false killer whales in the study area feed on large demersal fish but also on smaller schooling coastal species. Results suggest that, despite their scarcity, false killer whales do occur with some regularity in the coastal waters off northern New Zealand. The findings further highlight the need for greater sampling and accurate record keeping of stranded individuals, which combined have the potential to greatly improve our knowledge of this rarely sighted species.

Chapter Three

Site-fidelity, association patterns and preliminary population parameters



False killer whales off northeastern New Zealand showing some distinctively marked dorsal fins.
Photo Jochen Zaeschmar.

3.1 Introduction

The understanding of a species' population parameters (*e.g.* abundance, survival, movement patterns, site fidelity and group structure) in a given area is a crucial aspect of effective management. Such knowledge provides a benchmark against which changes can be measured in order to assess the population's health and devise appropriate management strategies. For many species, such knowledge is generated by the recognition of individuals and the subsequent analysis of their spatial and temporal occurrence and association patterns. Individuals can be identified by applying an artificial mark or by using natural features, with both techniques extensively applied in a wide range of taxa (*e.g.* Speed *et al.* 2007).

Artificial marking, by means of banding, colouring, branding or tagging remains prevalent in a wide range of taxa and continues to yield valuable information about the respective populations (*e.g.* Donehower and Bird 2005, McMahon *et al.* 2006, Hammerschalg *et al.* 2011, Saraux *et al.* 2011). However, the technique presents some potential limitations as marks may be lost over time or retrieved marks may not be reported (Speed *et al.* 2007). Additionally, the application and/or the wearing of the mark may alter the individual's subsequent behaviour, thereby introducing bias, which can be difficult to quantify (*e.g.* Murray and Fuller 2000, Walker *et al.* 2012).

The effects on the individual may range from benign to severe and appear to differ between species and techniques (*e.g.* Hill and Talent 1990, Jennings *et al.* 1991, Saraux *et al.* 2011, Walker *et al.* 2012). For example, zebra finches (*Poephila guttata*) showed differences in mating preference based on the colours of a potential mate's leg bands (Burley *et al.* 1982), while the overall life-span of meadow voles (*Microtus pennsylvanicus*) was reduced following the commonly used mark technique of toe-clipping (Pavone and Boonstra 1985). In cetaceans, the use of artificial marks to identify individuals was commonly used during commercial whaling operations (*i.e.* Omura and Ohsumi 1964) and infrequently during the earlier days of whale and dolphin studies (*i.e.* Wells and Scott 1990). The potential limitations together with the associated ethical issues have led to the increased use of animals' individual natural markings in order to study certain populations, ranging from badgers (*Meles meles*, Dixon 2003), to snow leopards (*Uncia uncia*, Jackson *et al.* 2006), whale sharks (*Rhincodon typus*, Holmberg *et al.* 2009) and marine mammals as diverse as harbour seals (*Phoca vitulina*, Thompson and Wheeler 2008), Hector's dolphins (*Cephalorhynchus hectori*, Webster *et al.* 2010), long-finned pilot whales (*Globicephala melas*, Auger-Mété and Whitehead 2007) or blue whales (*Balaenoptera musculus*, Olson 2008).

Photo-identification (photo-id) of cetaceans, based on each individual's unique markings, provides a non-invasive method of obtaining information on demographic parameters (Würsig and Jefferson 1990). The practice evolved in the 1970s, initially focusing on species that were easily accessible and displayed distinguishable natural markings that could be photographed without excessive effort such as the dorsal fins of common bottlenose dolphins (*Tursiops truncatus*, Shane and Wells 1986), the dorsal fins, saddle and eye patches of killer whales (*Orcinus orca*, Bigg *et al.* 1983), the tail flukes of humpback whales (*Megaptera novaeangliae*, Katona *et al.* 1979) or the callosities patterns of southern right whales (*Eubalaena australis*, Payne *et al.* 1983). To date, the technique is used on a wide range of odontocetes and mystecetes.

The initial application or identification and subsequent recapture of natural or artificial markings are known as mark-recapture. The method is commonly used to estimate a species' population parameters in a given area (Jolly 1965, Seber 1965). The ratio of marked to unmarked individuals is then calculated to estimate population abundance. In the present study, photo-id was used as a method of mark-recapture. Capture, marking and recapture are achieved by producing photographs that show the natural individual markings of each animal over time. This method relies heavily on the correct initial identification and the subsequent successful matching to the corresponding individual upon recapture. A number of factors such as poor quality images, lack of marks or changes in natural markings over time can affect correct recapture and considerably skew resulting population estimates (*e.g.* Hammond *et al.* 1990). Mark-recapture relies on a series of specific assumptions that need to be met in order to reduce bias (*e.g.* Seber 1982). These commonly include factors such as equal probability of capture and survival, no loss of marks and no behavioural response to the marking process. Finally, the type of analytical model needs to be chosen, based on whether the population to be estimated is considered demographically and geographically open or closed during the sampling period (Seber 1986).

False killer whales (*Pseudorca crassidens*) can be successfully photo-identified by the notches on their dorsal fins, with individuals remaining recognizable over periods of more than 20 years (Baird *et al.* 2008). Much of our knowledge about the species' population structure is derived from the extensive and ongoing photo-id studies conducted in Hawaiian waters (*e.g.* Baird *et al.* 2003, Baird *et al.* 2005, Baird *et al.* 2008). More limited studies exist from Costa Rican (Acevedo-Gutiérrez *et al.* 1997) and West-African waters (*e.g.* Weir *et al.* 2013). These examples constitute the only known studies of demographics, based on the identification of individuals and suggest at least some degree of site fidelity in nearshore waters, together with extended social bonds between individuals. Little or no information about the species' demographics exist from other regions.

Additionally, photo-id can be used to identify anthropogenic threats to cetacean species and populations (e.g. Visser 2000, Kiszka *et al.* 2008, Bradford *et al.* 2009). False killer whales are known to interact with fisheries (e.g. Ramos-Cartelle and Mejuto 2007), with the decline in some populations directly attributed to such encounters (Baird *et al.* 2014a). Baird and Gorgone (2005) concluded that major dorsal fin disfigurement in Hawaiian false killer whales was most likely caused by interactions with long-line fisheries, which led to the assessment of the prevalence of such injuries as an indicator of the threat level faced by the local populations.

Due to the general lack of studies on false killer whales in New Zealand waters, the population dynamics of the species in the study area remain unknown. Despite this paucity of information, the species is currently listed as *Not Threatened* by the New Zealand Threat Classification System (Baker *et al.* 2010).

This chapter aims to present an initial assessment of site fidelity, movement patterns, population parameters (abundance and survival) and social structure of false killer whales observed off northeastern New Zealand. Results are based on the analysis of photo-id efforts carried out in the study area between 2005 and 2012 (refer also to Zaeschmar *et al.* 2014, Appendix D). Additionally, the existence and/or prevalence of injuries suspected to have been sustained during fishery interactions is assessed. Findings are compared to studies from other regions to provide the necessary baseline data and recommendations required for appropriate management of the species in New Zealand waters.

3.2 Methods

3.2.1 Study area

The study area encompasses an approximately 650 kilometer (km) stretch of the northeastern coast of New Zealand, containing five locations where ongoing whale-watch operations and/or research projects were carried out. These are: the Three Kings Islands (TKI), the Bay of Islands (BOI), Poor Knights Islands (PKI), the Hauraki Gulf (HG) and the Bay of Plenty (BOP). The area extends from the north of the Three Kings Islands (approximate position 33°46' S, 171°27' E) to East Cape (approximate position 37°05' S, 178°4' E). The study area and its five study locations are described in detail in Chapter Two.

3.2.2 Survey platforms and methods

Photo-id was carried out from a range of opportunistic (*e.g.* commercial tour boats) and research platforms. (Table 3.1) Tour vessels encountered false killer whales opportunistically during wildlife/marine tours throughout the study area, following a similar, asystematic survey methodology, which was dictated by factors such as prevailing weather conditions but also suspected areas of likely cetacean occurrence and/or sighting reports from other vessels. In contrast, the research vessels encountered the species during dedicated cetacean surveys undertaken when visibility was > 1 km and Beaufort sea-state was ≤ 3 , using a continuous scanning methodology (*e.g.* Mann 1999). Once detected, false killer whales were approached slowly, with the vessels moving line abreast to travelling groups or slowly approaching stationary groups from the side and slightly behind, following New Zealand Marine Mammal Protection Regulations (1992). Survey methods are described in detail in Chapter Two.

3.2.3. Photo-identification

Standard photo-id methods (*e.g.* Würsig and Jefferson 1990, Baird *et al.* 2008) were applied to identify individual false killer whales. A *capture* was defined as one or more useable images of an individual taken on an independent day. Primary identification features included notches on or adjacent to the leading or trailing edge of the dorsal fin in addition to other permanent distinguishing features such as dorsal fin disfigurement. Secondary features included scars as well as fresh subdermal wounds such as those presumed to be the result of cookie cutter shark bites (*Isistius* spp.). Only primary features were used to confirm matches, with secondary features used only as an aid to identification. Dorsal fin images were graded according to the likelihood of successful recapture and matching (Table 3.2).

The quality of each image was assessed by its focus, contrast and the angle of the fin relative to the frame and graded on a scale of 1 to 4, with 1 being *excellent*, 2 being *good*, 3 being *fair* and 4 being *poor*. The best photograph obtained of an individual during an encounter was used for matching. The distinctiveness of each dorsal fin was graded on a similar scale of 1 to 4, with 1 being *very distinctive*, 2 being *distinctive*, 3 being *slightly distinctive* and 4 being *not distinctive* (Table 3.3). Only *very distinctive* and *distinctive* individuals and images of *excellent* or *good* quality were included in the analysis.

Table 3.1. Tour and research platforms used during the 15 photo-identification encounters off northeastern New Zealand between 2005 and 2012.

Location	Vessel name	Number of records	Operating season	Period of operation	Nature of operation	Vessel type	Length	Propulsion	Approx. observer eye height
Bay of Islands	<i>Manawanui</i>	5	Oct. - May	Since 2000	Tour boat	Steel sailing ketch	22 m	120 hp	4 m
	<i>Discovery V</i>	2	Oct. - May	2008 - 2012	Tour boat	Aluminium power catamaran	23 m	Twin 750 hp engines	7 m
	<i>Orca Research</i>	1	Year round	Since 1994	Research	Rigid inflatable	6.3 m	150 hp outboard	2 m
Three Kings Islands	<i>Cascade</i>	2	Apr. - May	2008	Tour boat	Fibreglass mono hull	16 m	Twin 435 hp	3 m
Poor Knights Islands	<i>Mazurka</i>	1	Year round	Since 2006	Tour boat	Steel mono hull	14 m	n/a	3.5 m
Hauraki Gulf	<i>Te Epiwhani</i>	1	Year round	2010-2012	Research	Aluminium mono hull	5.5 m	90 hp outboard	2 m
	<i>Dolphin Explorer</i>	1	Year round	Since 2000	Tour boat	Aluminium catamaran	20 m	Twin 350 hp	5 m
Bay of Plenty	<i>Aronui Moana</i>	2	Year round	2010 - 2012	Research	Aluminium mono hull	5.5 m	hp outboard	2 m
	<i>Guardian</i>	1	Oct. - May	Since 2006	Tour boat	Fibreglass catamaran	15 m	Twin 375 hp	4 m

Table 3.2. Grading system and assessment criteria applied for determining image quality used for photo-identification of false killer whales off northeastern New Zealand. The quality of each image was assessed by its focus, contrast and angle of the fin relative to the frame. Only images of *excellent* or *good* quality of were included in the analysis.









Image quality grading	Assessment criteria
1 (<i>excellent</i>)	 <p>All quality criteria are met: sharp focus with clear contrast and taken at an angle that allowed a clear profile of the dorsal fin's leading and trail edge.</p>
2 (<i>good</i>)	 <p>One of the quality criteria was compromised but the information content remained intact, allowing for the identification of <i>very distinctive</i> and <i>distinctive</i> individuals.</p>
3 (<i>fair</i>)	 <p>Two or more quality criteria were compromised allowing only for identification of <i>very distinctive</i> individuals.</p>
4 (<i>poor</i>)	 <p>One or more quality criteria were compromised to the point that successful identification of the individual was not possible.</p>

Table 3.3. Grading system and assessment criteria applied for determining dorsal fin distinctiveness used for photo-identification of false killer whales off northeastern New Zealand. The distinctiveness of each dorsal fin was assessed by the size and number of notches on the leading or trailing edges of the fin. Only *very distinctive* and *distinctive* individuals were included in the analysis.

Dorsal fin distinctiveness grading	Assessment criteria
1 (<i>very distinctive</i>)	 <p data-bbox="1066 526 1404 674">Multiple notches, including large notches and could be identified from photos of all quality categories.</p>
2 (<i>distinctive</i>)	 <p data-bbox="1066 887 1404 994">Multiple notches and could be identified from <i>excellent</i>, <i>good</i> and <i>fair</i> photographs.</p>
3 (<i>slightly distinctive</i>)	 <p data-bbox="1066 1234 1404 1341">Few notches and could only be identified from <i>excellent</i> or <i>good</i> photographs.</p>
4 (<i>not distinctive</i>)	 <p data-bbox="1066 1581 1404 1881">Clean fins (<i>e.g.</i> no notches or other permanent distinguishing features) or showed notches that could only be seen in <i>excellent</i> images within an encounter but unlikely between encounters.</p>

Each new image was carefully examined to avoid false positives (the matching of two or more distinct animals to the same catalogue number) and false negatives (the same individual being assigned multiple catalogue numbers) (Hammond *et al.* 1990, Wilson *et al.* 1999, Berghan *et al.* 2008). All matches were confirmed by at least two experienced researchers. Mark changes (*e.g.* the acquisition of one or multiple new notches or a change in notch size or shape) were identified based on two or more notches being the same (the same shape and the relative positioning on the fin).

Successful photo-ids were entered into the New Zealand False Killer Whale Identification Catalogue (NZFKWC, Zaeschmar unpubl. data⁶) respectively. As images were collated from a number of different sources, homogeneity of data collection could not be ensured. For example, it could not be determined if photographs taken aboard tour boats were taken randomly or if they were biased towards individuals displaying greater levels of interaction and/or more interesting behaviours. Likewise, the time spent with false killer whales was often not adequate to successfully sample the entire group due to time, operating limit or weather constraints. Consequently, it has to be assumed that at least some of the groups were not sampled completely and/or at random. To evaluate this issue, possible relationships between encounter duration, the number of images taken and the number of individuals identified were analysed.

3.2.4 Social network

A social network diagram of false killer whales photo-identified in the study area was produced using the program *Netdraw 2.123*⁷. A spring embedded layout was selected, placing more connected nodes at the centre of the diagram, while those with fewer connections were placed around the periphery. The existence of social clusters was tested in *Socprog 2.4*.

3.2.5 Minimum home ranges

To assess differences in individual false killer whales' ranges within the study area (following Baird *et al.* 2008), the distances between all possible combinations of sighting locations were calculated using ArcGIS 9.3 (ESRI). Furthermore, for each individual sighted on more than one occasion, the distances between all locations where the individual had

⁶ The NZFKWC is curated by J.R. Zaeschmar, Coastal-Marine Research Group, Institute of Natural and Mathematical Sciences, Massey University, Private Bag 102904, North Shore, Auckland 0745, New Zealand; E-mail: jzaeschmar@hotmail.com

⁷ Borgatti, S.P., 2002. NetDraw Software for Network Visualization. Analytic Technologies: Lexington, KY Available from analytictech.com/Netdraw

been sighted were measured. Distances between all possible combinations of sighting locations were then compared to the median distances between all locations where each individual had actually been sighted.

3.2.6 Dorsal fin disfigurement

In order to assess the existence and/or extent of false killer whale interactions with fisheries in the study area, dorsal fins were also examined for injuries consistent with those caused by fishing gear such as monofilament lines and/or nets. Following Baird and Gorgone (2005), major dorsal fin disfigurement was classified as the dorsal fin being bent over completely at the base or missing altogether.

3.2.7 Proportion of marked individuals, rate of mark change and rate of discovery of previously uncatalogued individuals

The proportion of individuals with identifiable notches on the trailing or leading edges of their dorsal fins was assessed by counting the number of marked and unmarked individuals on photographs of sufficient quality (*excellent* or *good*). One encounter (PKI 2010) was omitted from the mark rate calculations due to the small number of photo-id of sufficient quality ($n = 3$). Following Baird *et al.* (2008), the rate of mark change was calculated by dividing the sum of all resighting intervals by the minimum number of mark changes. The rate of discovery of marked individuals was calculated from the cumulative number of identified whales in relation to the number of newly identified individuals using *Socprog* 2.4⁸ (Whitehead 2009).

3.2.8 Mark-recapture

As deaths, births, emigration and/or immigration could not be ruled out over the sampling period (seven years), a Cormack-Jolly-Seber framework was implemented to evaluate potential violations of mark-recapture assumptions, which are as follows:

1. No loss of marks during the sample period

Previous research indicates that false killer whales may remain identifiable over decades (Baird *et al.* 2009). The rate of mark change was evaluated and shown to be low, which is consistent with studies from other regions (Baird *et al.* 2008). As such, individuals photo-identified in the present study area remained identifiable throughout the duration of the study. While it cannot be ruled out that some individuals may acquire mark changes that

⁸ Dalhousie University, Halifax, Nova Scotia, Canada. Available from myweb.dal.ca/~hwhitehe/social.htm

render them unrecognizable, the overall low rate of mark change and the restriction of the analysis to only *very distinctive* and *distinctive* individuals should minimise the risk of mismatches (Baird *et al.* 2008). Consequently, permanent marks on the dorsal fins of false killer whales are, therefore, considered reliable indicators of an individual's identity.

2. Sampling is instantaneous

Given that photo-id does not require any actual marking of individuals, no capture or handling of individuals is necessary. Sampling was therefore considered instantaneous.

3. No behavioural response to the marking process

Photo-id itself is not expected to affect the behaviour of false killer whales because the non-invasive nature of the technique is unlikely to elicit a capture response. However, attraction to and/or avoidance of observation platforms or heterogeneity in residency patterns (Tezanos-Pinto and Baker 2012) has to be considered. To evaluate potential behavioural effects that would suggest a response to capture (*e.g.* 'trap-happy' or 'trap-shy'), TEST 2.CT was implemented in U-CARE version 2.2 (Choquet *et al.* 2005).

4. The homogeneity of capture is maintained throughout the sampling period

Homogeneity of capture is based on the assumption that all individuals (*e.g.* marked, unmarked, male, female *etc.*) are equally likely to be captured in the study area. Unequal capture probability may result in heterogeneity of individual capture, which may be occurring if individuals spend less time in the area, are easier to photograph, avoid or prefer boats and/or associates (*e.g.* mother calf pairs, frequent companions) (Wilson *et al.* 1999).

5. Equal probability of survival

Every sampled individual has the same probability of survival between capture and next recapture (Hammond 1986). In this context, survival refers to 'apparent survival' because deaths are confounded by emigrations and births by immigrations. Variations in apparent survival are therefore changes in either mortality or emigration (or both). TEST 3.SR and 3 SM were implemented in U-CARE to determine if there was an excess of 'transient' individuals (*e.g.* individuals only sighted once) and to examine the potential effects of capture on survival, respectively (Choquet *et al.* 2005).

Since an open model was chosen, using only distinctive false killer whales, it is important to emphasise that the abundance estimate produced herein refers to the number of distinctive individuals present in the study area during the sample period. No assumptions are made as

to how this estimate relates to the wider false killer whale population. It is, therefore, referred to as the apparent abundance estimate.

The Schwarz and Arnason ‘superpopulation’ parameterization of the Jolly-Seber model (*e.g.* POPAN; Crosbie and Manly 1981, Schwarz and Arnason 1996) was employed, with each encounter used as a session to gain an understanding of apparent survival rates and apparent abundance of the whales that have visited the area (*e.g.* users and visitors; Williams *et al.* 2002).

3.2.8.1 POPAN model

A superpopulation (POPAN) approach was applied in MARK with data pooled by years. This approach provides estimates of apparent survival rates of false killer whales off northeastern New Zealand, apparent annual abundance and the apparent abundance of all whales that visited the study area during the sampling period. This approach is based on a re-parameterization of the Jolly-Seber (JS) model with an additional parameter N_{super} that denotes the size of the ‘superpopulation’. The intervals between encounters were specified in decimal years to obtain consistent, per annum estimates of apparent survival. The model estimates the apparent survival probability (ϕ) and probability of entry (β) between encounters, the capture probability (p), and apparent abundance (N) of false killer whales for each encounter. Models were considered with constant (.) and temporal variation (t) in capture probabilities between years. A constraint was added to the first two and the last two capture probabilities to provide parameter identifiability for all models (Cooch and White 2011). The average duration of encounters per year was modelled in the design matrix to examine if average encounter duration affected capture probabilities.

3.2.8.2 Goodness of fit tests and model selection

The dataset was pooled by years and analysed in a Cormack-Jolly-Seber (CJS) framework in order to estimate the variance inflation factor (\hat{c}) and to carry out goodness of fit tests. Median \hat{c} was estimated in MARK (White and Burnham 1999, Cooch and White 2011). Where median \hat{c} was greater than one (indicative of overdispersion in the data), a \hat{c} was incorporated into a QAIC_c statistic and used instead of AIC_c in model selection (Quasi-likelihood Akaike Information Criterion; Burnham and Anderson 2002).

Goodness of fit tests (TEST 2 and TEST 3) were run in U-CARE V 2.2 (Choquet *et al.* 2005) to evaluate potential violations of assumptions for both data sets. A significant result in TEST 2 indicates that capture probabilities differ among individuals (heterogeneity). TEST 2 can be further partitioned into TEST 2.CT, which examines whether there is a

behavioural response to the first capture (trap-avoidance statistic $z > 0$, trap-happy statistic $z < 0$) and TEST 2.CL, which examines whether there is variation in the time between re-encounters for captured and not captured (but known to be alive) individuals. A significant result in TEST 2.CL indicates that the trap effect lasts for more than one interval. TEST 3 evaluates the assumption that all individually identifiable whales have the same probability of survival between sampling occasions. TEST 3 is partitioned into 2 additional tests: TEST 3.SR incorporates a statistic for transience, with a significant result ($z > 0$; $P < 0.05$) suggesting a transience effect (*e.g.* whales sighted only once during the course of the study more often than expected), whereas TEST 3.SM examines whether there is an effect of capture on survival (Choquet *et al.* 2005).

3.3 Results

Of the 47 false killer whale encounters documented in the study area between March 1995 and February 2012 (described in Chapter Two), *excellent* or *good* photo-id images of dorsal fins were obtained during 31.9% of encounters ($n = 15$). The majority of these encounters were in BOI (2005-2007, 2009-2010, 46.7%, $n = 7$), followed by BOP (2009 and 2012, 20.0%, $n = 3$), TKI (2008, 13.3%, $n = 2$), HG (2011, 13.3%, $n = 2$) and PKI (2010, 6.7%, $n = 1$) (Figure 3.1, Table 3.4).

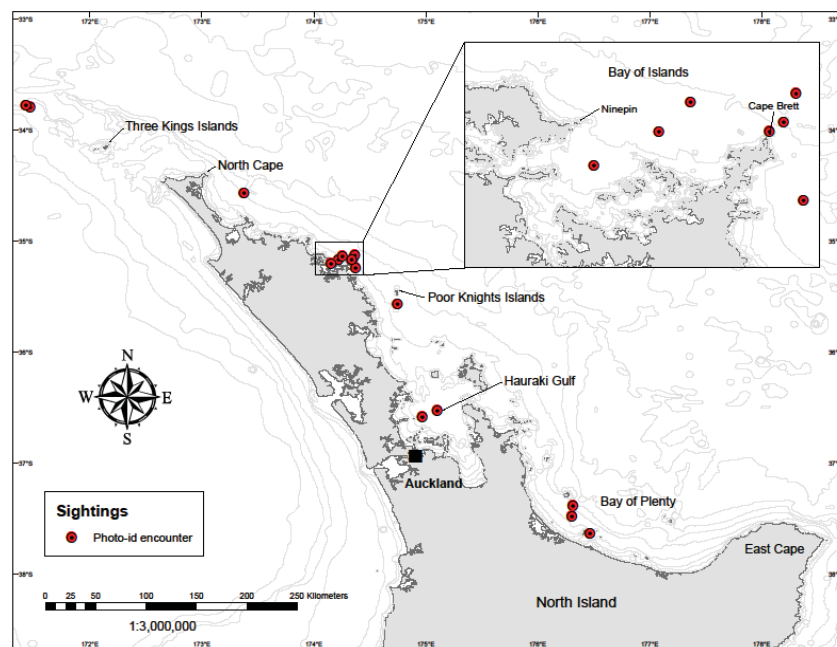


Figure 3.1. Study area showing the locations of the 15 false killer whale encounters (red circles) off northeastern New Zealand during which photo-identification was carried out between 2005 and 2012.

Table 3.4. Details of the 15 false killer whale encounters off northeastern New Zealand between 2005 and 2012 where photo-identification images of sufficient quality were collected. Area codes are Bay of Islands (BOI), Bay of Plenty (BOP), Hauraki Gulf (HG), Poor Knights Islands (PKI) and the Three Kings Islands (TKI). Observations were made from tour boats (T) and/or research vessels (R). All observations were of mixed-species groups of false killer whales (Pc) and bottlenose dolphins (Tt).

Date	Area	Vessel type	Group type	Total group size	Group size Pc	Group size Tt	Total ID photos taken	Marked individuals	Unmarked individuals	Total individuals identified	Percentage of estimated group size	Encounter duration in minutes (min)
2005-Feb-11	BOI	R	M	80	30	50	43	26	17	8	26.7	105
2007-Jan-11	BOI	R	M	100	50	50	176	134	42	34	70.0	210
2007-Apr-03	BOI	R	M	80	30	50	54	40	14	8	33.3	165
2007-Dec-28	BOI	R	M	150	50	100	164	120	44	33	66.0	225
2008-May-16	TKI	T	M	100	40	60	15	10	5	6	15.0	20
2008-May-17	TKI	T	M	100	40	60	40	35	5	10	25.0	15
2009-Feb-25	BOP	T	M	70	30	40	48	34	14	10	33.3	45
2009-Dec-21	BOI	R	M	110	50	50	230	189	41	46	92.0	180
2010-Mar-20	BOI	T	M	80	30	50	37	22	15	9	30.0	20
2010-Mar-25	BOI	T	M	65	60	5	20	14	6	3	5.0	120
2010-Apr-20	PKI	T	M	60	30	30	3	1	2	1	3.3	20
2011-Jan-20	HG	T	M	300	150	150	165	116	49	22	14.7	60
2011-Jan-25	HG	R	M	90	30	60	63	54	9	14	46.7	118
2012-Jan-18	BOP	R	M	170	30	200	94	69	25	20	66.7	120
2012-Feb-09	BOP	R	M	120	30	150	80	54	26	16	53.3	100

A total of 79 individuals were identified, 22.8% ($n = 18$) of which were considered only *slightly distinctive* resulting in 61 *very distinctive* or *distinctive* individuals included in the analysis. Using only *excellent* and *good* quality photographs, 226 identifications of these 61 individuals were made ($\bar{x} = 3.7$, $SD = 2.1$). The number of individuals identified in each encounter ranged from 1 to 41 whales ($\bar{x} = 14.7$, $SD = 11.1$, $n = 15$), while the mean group size for encounters with identified individuals was 49 ($SD = 31.6$, $n = 15$).

3.3.1 Resight rate

Of the 61 *very distinctive* and *distinctive* individuals, 88.5% ($n = 54$) were resighted, with 70.5% ($n = 43$) encountered on three or more occasions and two individuals observed on eight occasions (Table 3.5). Additionally, 85.2% ($n = 52$) were observed across years, with 8.2% ($n = 5$) documented in five different years between 2005 and 2012 (Figure 3.2). The highest number of resightings between any two encounters was 29 individuals (January 2007 and December 2009, both BOI). The shortest time frame between any two resightings of an individual was 5 days ($n = 4$). The longest time-frame between initial identification of an individual and its most recent resighting (disregarding sightings in between) was 2,551 d (*ca.* 7 years, $n = 4$, Figure 3.3).

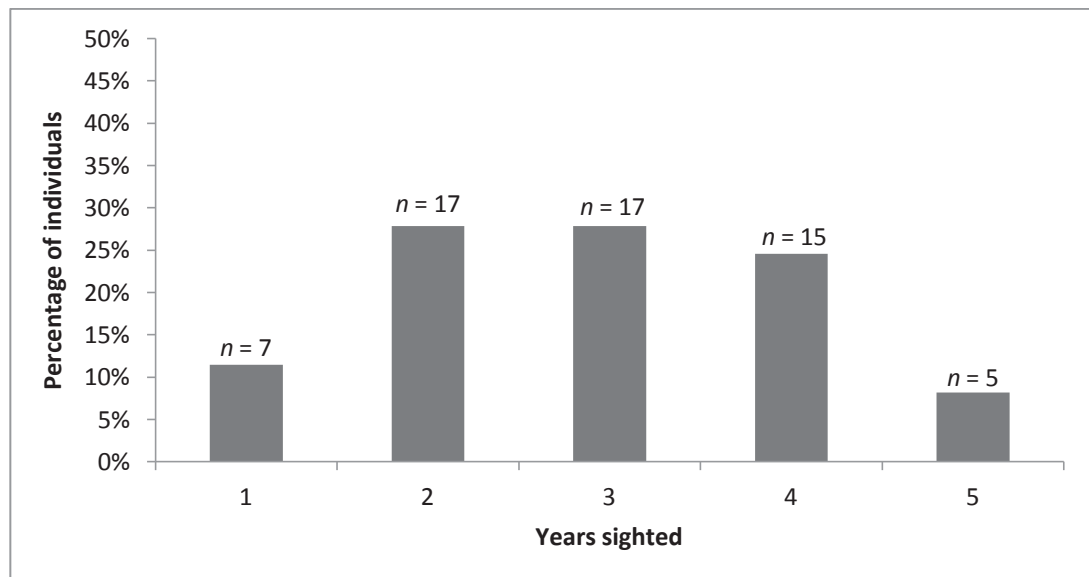


Figure 3.2. Number of different years in which identified individual false killer whales were observed off northeastern New Zealand between 2005 and 2012.

Distances between sighting locations ranged from < 1 km (BOI, 1074 days, $n = 29$) to *ca.* 650 km (TKI – BOP, 284 days, $n = 8$). The majority of individuals (77.0%, $n = 47$) was encountered in more than one of the five sighting locations within the study area, with 4.9% of individuals ($n = 3$) encountered in four of the five locations (Figure 3.4).

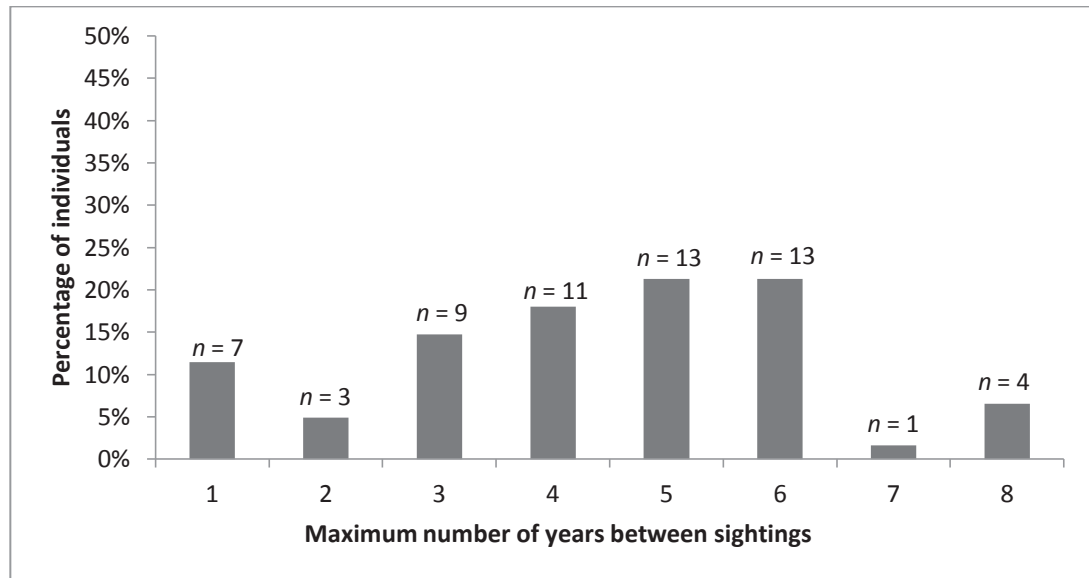


Figure 3.3. Maximum number of years between sightings of individual false killer whales photo-identified off northeastern New Zealand between 2005 and 2012.

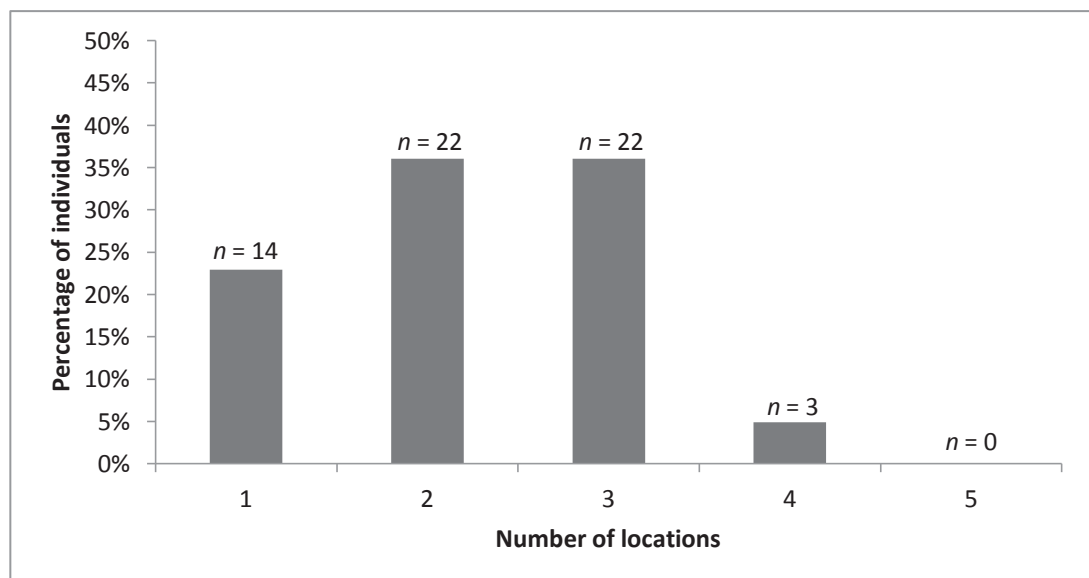


Figure 3.4. Number of individual false killer whales in relation to the number of different locations they were sighted in off northeastern New Zealand between 2005 and 2012.

Table 3.5. Photo-identified *very distinctive* and *distinctive* false killer whales encountered off northeastern New Zealand in Three Kings Islands (TKI), Bay of Islands (BOI), Poor Knights Islands (PKI), Hauraki Gulf (HG) and Bay of Plenty (BOP) showing movements between the different encounter locations and associations between individuals between 2005 and 2012.

Whale ID	Encounter date														
	16	17	11	11	03	28	21	20	25	20	20	25	25	18	09
	May	May	Feb	Jan	Apr	Dec	Dec	Mar	Mar	Apr	Jan	Jan	Feb	Jan	Feb
	2008	2008	2005	2007	2007	2007	2009	2010	2010	2010	2011	2011	2009	2012	2012
	Estimated group size														
	40	40	30	50	30	50	80	30	60	30	150	30	30	30	30
	Encounter location														
	TKI		BOI							PKI		HG		BOP	
NZ-Pc-001															
NZ-Pc-002															
NZ-Pc-003															
NZ-Pc-004															
NZ-Pc-005															
NZ-Pc-006															
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NZ-Pc-040															
NZ-Pc-041															
NZ-Pc-043															
NZ-Pc-047															
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NZ-Pc-068															
NZ-Pc-069															
NZ-Pc-070															

3.3.2 Social network

Of the 61 *very distinctive* and *distinctive* individuals photo-identified in the study area, all were linked by association in one large social network, albeit separated into two social clusters (Figure 3.5). This clustering was also apparent in the *Socprog* analysis (Figure 3.6).

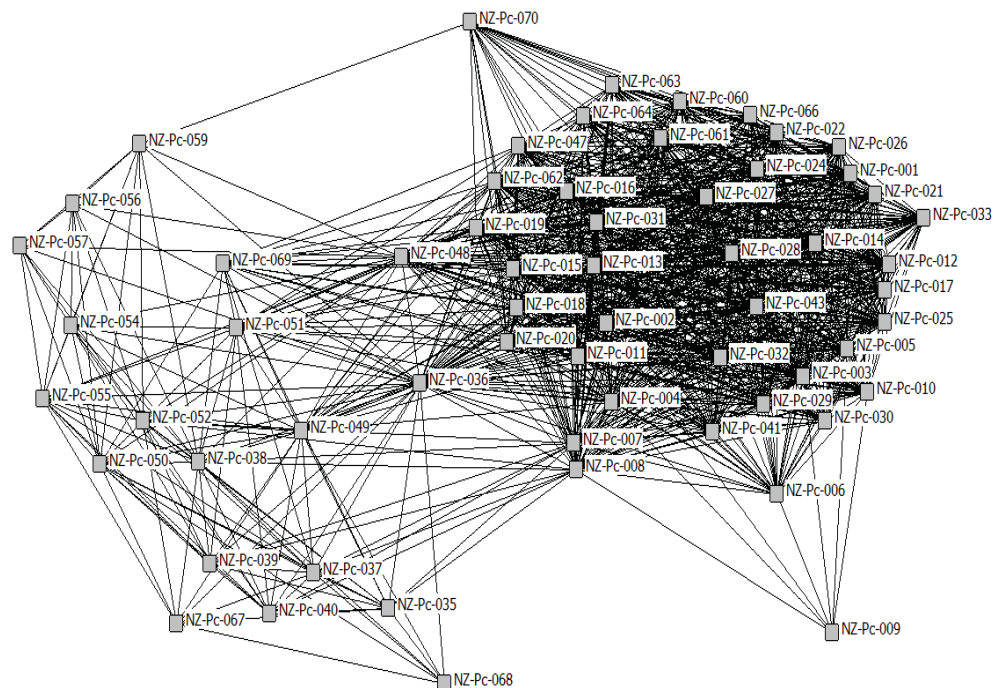


Figure 3.5. Social network diagram of 61 *very distinctive* or *distinctive* false killer whales photo-identified off northeastern New Zealand during 15 encounters between 2005 and 2012 using a spring embedded layout. Individual false killer whales with their corresponding catalogue number are represented by nodes. Only individuals with *excellent* or *good* quality photos are included. Note: Clustering suggests the existence of two social clusters.

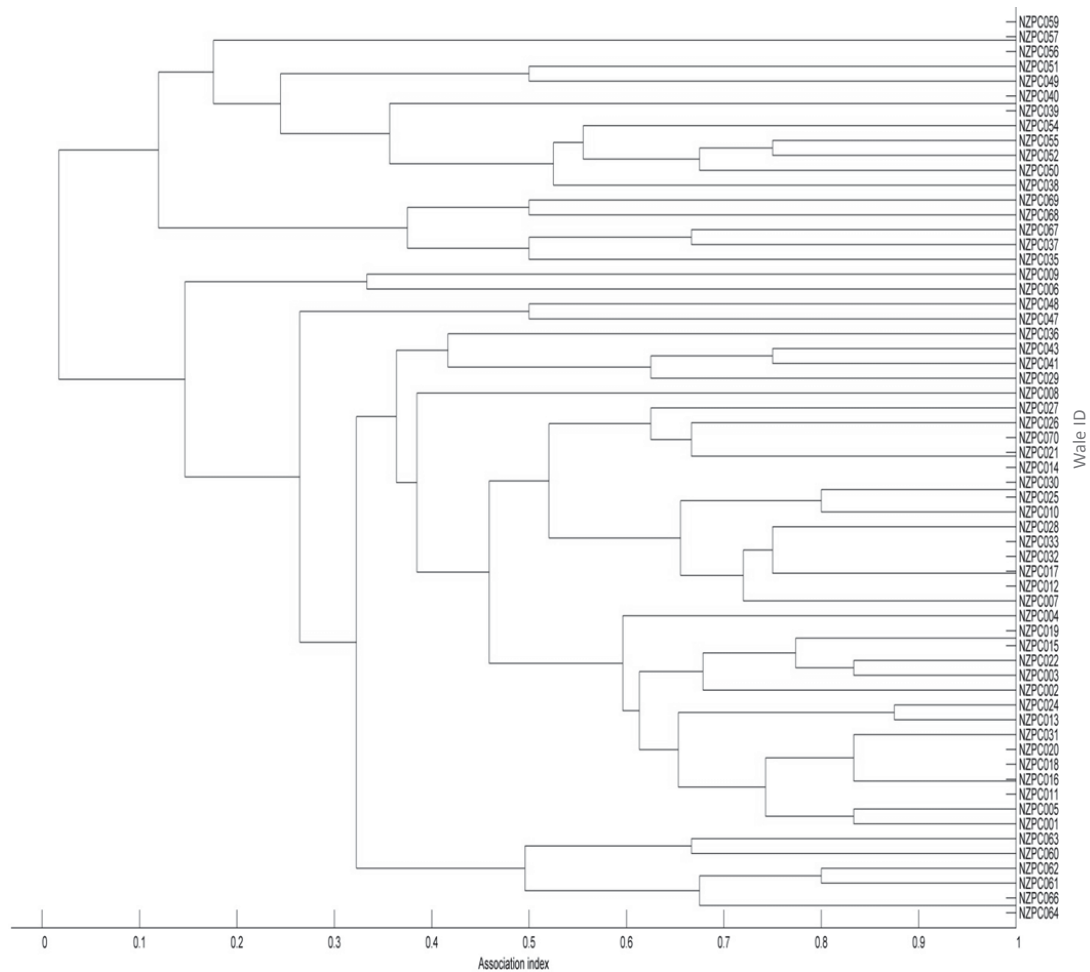


Figure 3.6. Dendrogram showing the association index of 61 *very distinctive* and *distinctive* false killer whales photo-identified off northeastern New Zealand between 2005 and 2012, indicating the existence of two distinct social clusters.

3.3.3 Minimum home ranges

The median distance among all possible combinations of sighting locations (the distance individuals could have travelled) was 210.9 km (range 1.8 - 647.2 km, SD = 167.3 $n = 105$). The median distance among all the sighting locations (the distance individuals did travel) of individuals encountered more than once ($n = 54$) was 188.2 km (range 4.3 - 647.2 km, SD = 149.6, $n = 396$).

3.3.4 Mark rate

The proportion of marked *vs.* unmarked individuals, excluding calves was assessed from images taken during 14 discrete encounters (Table 3.6). The proportion of marked individuals varied considerably between encounters, ranging from 59.5% to 87.6 ($\bar{x} = 72.7$, SD = 8.3, $n = 14$)

Table 3.6. False killer whale encounters ($n = 14$) off northeastern New Zealand between 2005 and 2012 used for mark rate assessment.

Date	Total marked	Total unmarked	Total photos	Total identified individuals	Mark rate (%)	Encounter duration (min.)	Group size estimate
11-Feb-2005	26	17	43	8	60.5	105	30
11-Jan-2007	134	42	176	34	76.1	210	50
3-Apr-2007	40	14	54	8	74.1	65	30
28-Dec-2007	120	44	164	33	73.2	225	50
16-May-2008	10	5	15	6	66.7	20	40
17-May-2008	35	5	40	10	87.5	15	40
25-Feb-2009	34	14	48	10	70.8	45	30
21-Dec-2009	189	41	230	46	82.2	180	80
20-Mar-2010	22	15	37	9	59.5	20	50
25-Mar-2010	14	6	20	3	70.0	120	60
20-Jan-2011	116	49	165	22	70.3	60	150
25-Jan-2011	54	9	63	14	85.7	65	30
18-Jan-2012	69	25	94	20	73.4	120	30
9-Feb-2012	54	26	80	16	67.5	100	30
Mean	65.5	22.3	87.8	17.1	72.7	96.4	50

3.3.4.1 Mark change

At total of 24 changes in either the number or shape of notches, were documented on 17 individuals between 2005 and 2012 (Figure 3.7). The sum of all resighting intervals was 196.9 years, resulting in one mark change every 8.2 years.

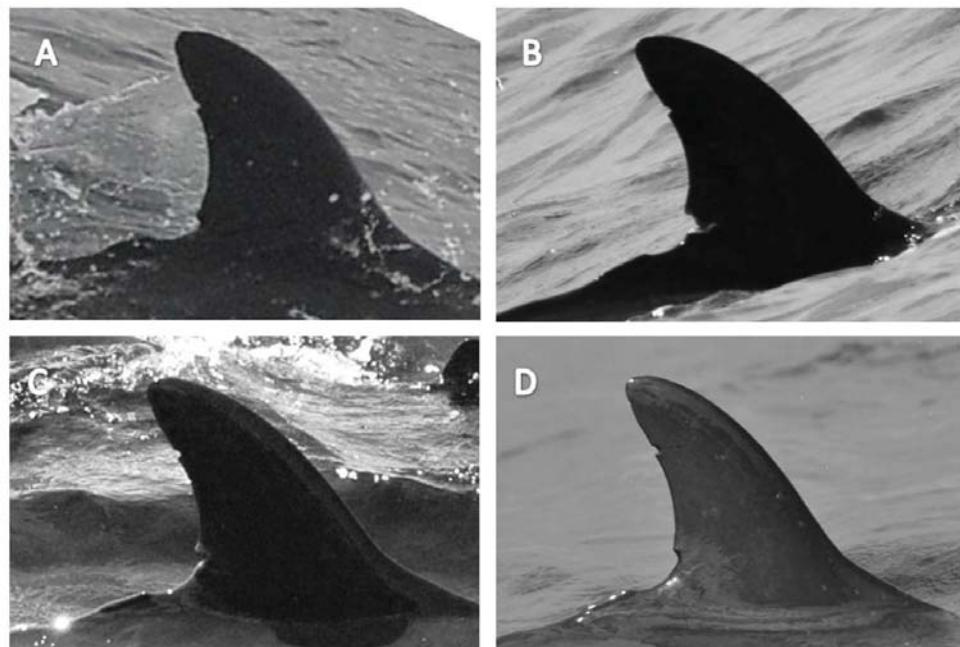


Figure 3.7. Examples of mark change of a false killer whale photo-identified off northeastern New Zealand over a five year period. *Distinctive* individual NZ-Pc-018 first photographed on January 11, 2007 (A, photo quality *good*). Resighted on December 20, 2009 with freshly wounded dorsal fin (B, photo quality *good*), on January 25, 2011 with healing fin (C, photo quality *good*) and on January 18, 2012 with completely healed fin (D, photo quality *excellent*).

3.3.5 Dorsal fin disfigurement

Dorsal fin disfigurement was observed in 3.3% ($n = 2$) of photo-identified individuals (Figure 3.8). However, only one individual (NZ-Pc-025) met the criteria for disfigurement considered consistent with long-line fisheries as detailed by Baird and Gorgone (2005). Both individuals showed dorsal fin disfigurements when first photo-identified in 2005 (NZ-Pc-005) and 2007 (NZ-Pc-025). There have been no new fishery related injuries observed on previously identified individuals since the beginning of the photo-id study in 2005.

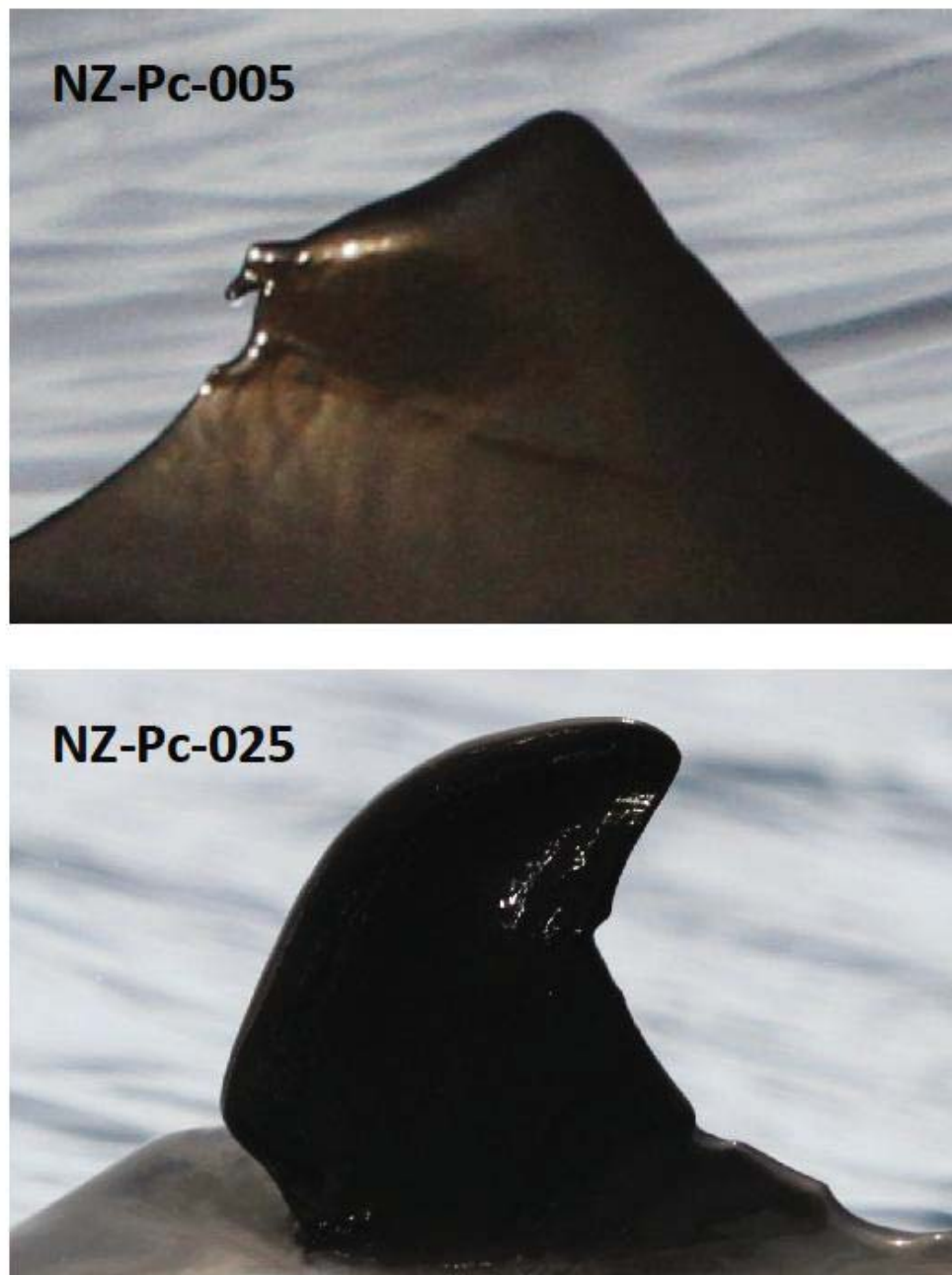


Figure 3.8. Examples of dorsal fin disfigurement, presumed to be the result of fishery interactions NZ-Pc 005 (top) and NZ-Pc 025 (bottom). Photos © Jochen Zaeschmar.

3.3.6 Rate of discovery of previously uncatalogued individuals

Overall, the proportion of new identifications decreased as the rate of resightings increased (Figure 3.9). While every encounter after the initial observation included previously catalogued individuals, previously unidentified individuals were only captured during 53.3% of encounters ($n = 8$). The discovery curve appears to have plateaued, indicating that most individuals that frequented the study area during the sampling period have been photo-identified (Figure 3.10).

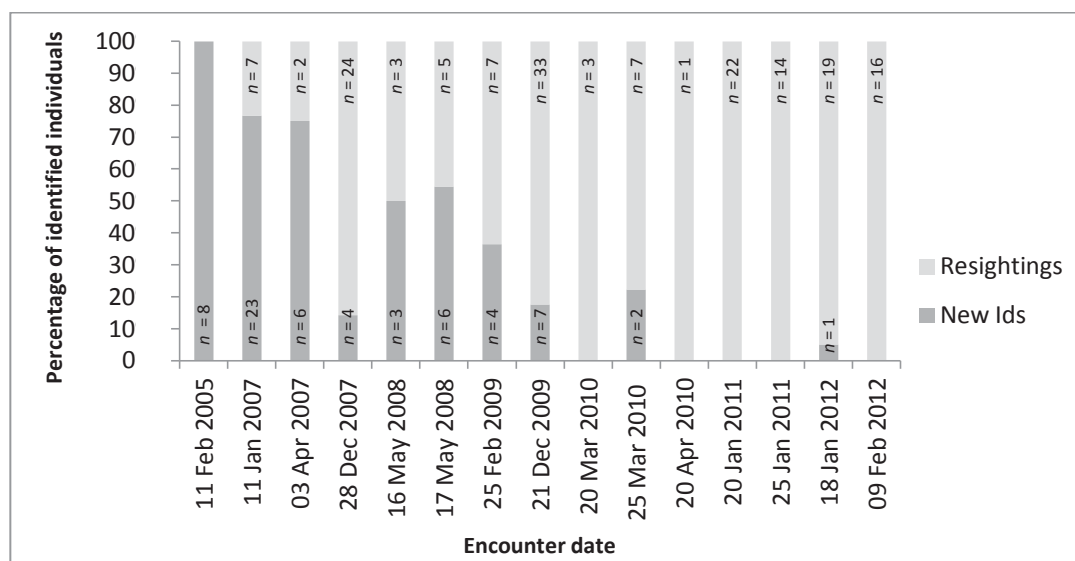


Figure 3.9. Percentages of newly (New IDs) and previously (Resightings) photo-identified false killer whales per encounter. Recorded off northeastern New Zealand between 2005 and 2012.

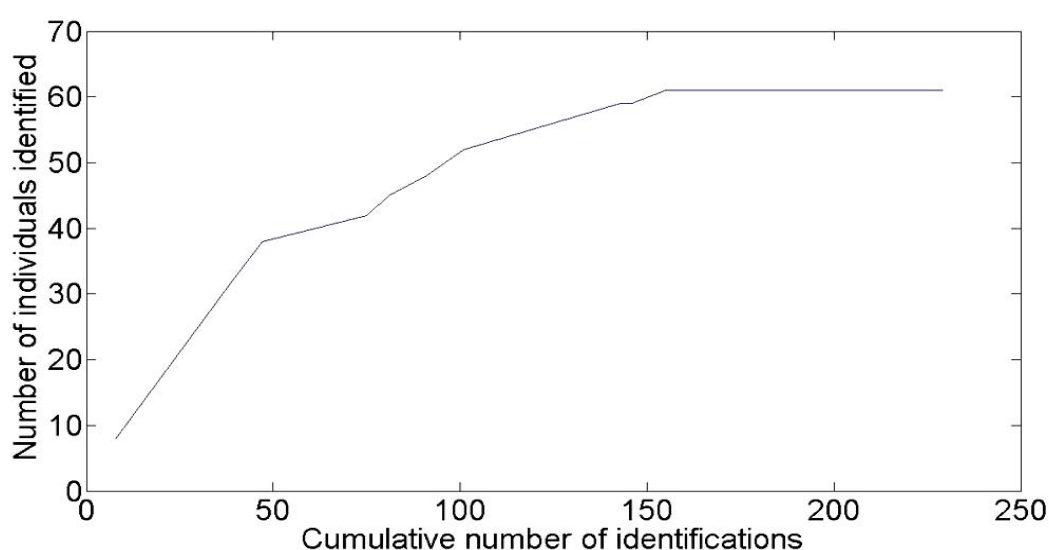


Figure 3.10. Discovery curve showing the cumulative number of identifications in relation to the number of identified individuals. Recorded off northeastern New Zealand between 2005 and 2012.

3.3.7 Apparent abundance and apparent survival

3.3.7.1 Goodness of fit test

Results conducted in U-CARE suggested a ‘trap-shy effect’ (test 2.CT); while all other test were not significant (Table 3.7).

Table 3.7. Goodness of Fit tests run in U-CARE for false killer whale photo-identification data collected off northeastern New Zealand between 2005 and 2012, including the results of the global test (Tests 2+3). Abbreviation d.f. = degrees of freedom.

Tests	2.CL	2.CT	3.SM	3.SR
Statistics		3.741		-0.249
P value	0.494	0.001	0.959	0.881
X ²	1.408	15.81	0.63	1.181
df	2	3	4	
Global Test	X ² = 19.03 d.f. = 13 P = 0.122			

3.3.7.2 Mark-recapture models

Median c-hat was estimated at 2.32 and models were adjusted for this value. There were two competing models. The first model incorporated constant survival, time-varying capture probability and time-varying probability of entry. The second model incorporated constant survival, time varying capture probability incorporating encounter duration and time-varying probability of entry (Table 3.8).

Table 3.8. Model selection for sighting data of false killer whales collected from 2005-2012 for the POPAN annual data. Abbreviations: apparent survival (ϕ), capture probability (p) and probability of entry (β). Constrained capture probability ($k = k-1$). Qdev = deviance, t = variation between years. The lowest QAICc value represents the model that has the most support from the data. NP denotes the number of parameters, ML = model likelihood.

Model	QAICc	Delta QAICc	AICc Weights	ML	NP	QDev
$\phi(.) p(t, k = k-1) \beta(t)$	186.0502	0	0.74401	1	14	-34.9633
$\phi(.) p(t^* \text{enc dur hr}) \beta(t)$	188.3776	2.3274	0.23238	0.3123	15	-34.9633
$\phi(t) p(t, k = k-1) \beta(t)$	193.5755	7.5253	0.01728	0.0232	19	-39.3251
$\phi(t) p(t) \beta(t)$	195.8627	9.8125	0.00551	0.0074	20	-39.4924
$\phi(.) p(\text{enc dur hr}) \beta(t)$	199.8142	13.764	0.00076	0.001	15	-23.5268
$\phi(t) p(t, k = k-1) \beta(.)$	204.6323	18.5821	0.00007	0.0001	14	-16.3812
$\phi(t) p(.) \beta(t)$	220.5828	34.5326	0	0	14	-0.4307

3.3.7.3 Population parameters

Model averaging was conducted to estimate population parameters (apparent survival, apparent annual abundance, probability of entry and the size of the ‘superpopulation’). Apparent survival was estimated at 0.95 (SE = 0.034, CI = 0.821-0.987). Apparent annual

abundance ranged from a low of 51 marked whales in 2007 to a high of 74 marked whales in 2008 (Figure 3.11). The probability of entry ranged from 0.024 (SE = 0.058, CI = 0.0002 - 0.767) in 2010 to a high of 0.53 (SE = 0.092, CI = 0.356-0.696) in 2007. Overall, the total number of marked false killer whales that were sighted in the study area from 2005 to 2012 was estimated at 81 (SE = 3.038, CI = 75.203 - 87.114). Based on the mean mark rate of 72.7% (range 59.5-87.5%, SE = 2.23, $n = 14$), the superpopulation of whales observed in the study area was estimated at 111 individuals (range 92 – 136, CI = 101.173 – 123.341).

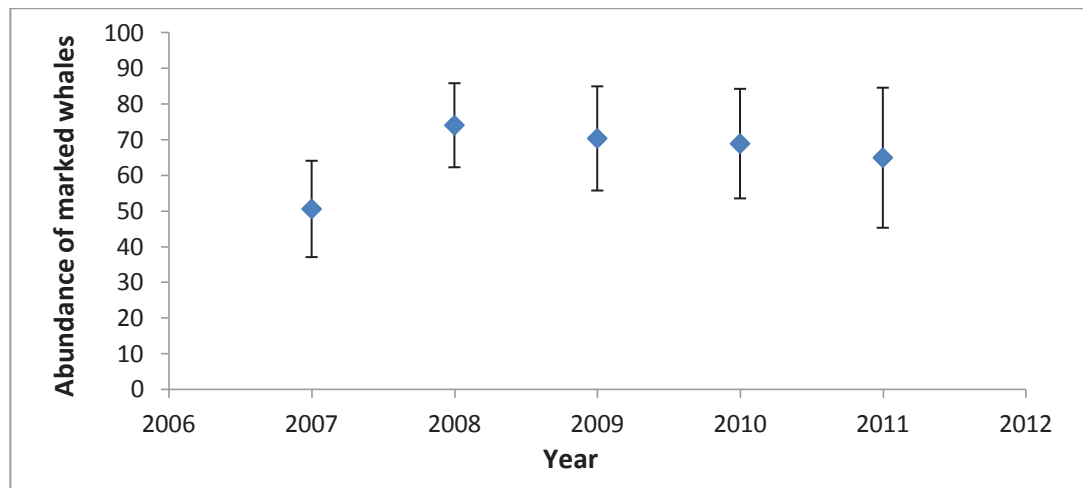


Figure 3.11. Apparent abundance of marked false killer whales photo-identified off northeastern New Zealand between 2005 and 2012 estimated with POPAN (model averaging). Note: apparent abundance estimates for 2005 and 2012 presented too large confidence intervals and therefore were deemed unreliable.

3.4 Discussion

3.4.1 Resight rate

While false killer whales may initially appear to be infrequent visitors within New Zealand waters, photo-id data presented here suggest recurrent use of the study area by the majority of identified animals. False killer whales showed both short- and long-term site fidelity, with most individuals observed repeatedly in the same area within a given year and as much as seven years apart (see also Zaeschmar *et al.* 2014, Appendix D). This level of site fidelity is higher than would be expected from a presumed oceanic species and may yet be underestimated due to the incomplete sampling of some groups. However, these findings are consistent with photo-id from Hawai'i (Baird *et al.* 2008) and Costa Rica (Acevedo-Gutiérrez *et al.* 1997).

3.4.2 Social network

Known individuals exhibited evidence of long-term associations, in many cases lasting years and spanning hundreds of kilometers. All individuals photo-identified in the study area so far, are linked by association in a single social network. Results further suggest the existence of two social clusters within this social network, similar to the clustered social structure identified within the Hawaiian insular false killer whale population (Baird *et al.* 2012). However, given the likelihood that not all individuals were photo-identified in some of the encounters during the present study, the apparent clustering may result from incomplete capture of all individuals present in the groups observed.

3.4.3 Minimum home ranges

The majority of identified individuals were resighted in multiple locations within the study area, with distances of up to 650 km documented between sightings. There was little discrepancy between the distances individuals could have travelled and did actually travel, suggesting no obvious individual differences in the use of the study area despite the apparent clustering. Conversely, satellite tagging of individuals in Hawaiian waters revealed differing home-ranges between individuals within the island-associated population (Baird *et al.* 2012). However, the findings of the present study may be biased by the incomplete sampling of groups. Furthermore, it cannot be ruled out that home ranges of individuals overlap in the study area, with possible differences only detectable at larger scales.

3.4.4 Mark rate

The high overall mark rate (72.7%) and the low mark change rate of 8.2 years are consistent with that observed in Hawaiian waters (77%, 6.9-8.8 years, Baird *et al.* 2005, 2008), which further support photo-id as a viable method to monitor population dynamics of false killer whales. This success is despite the low encounter rate and the resulting long sighting intervals in the study area. However, differences in the methods of mark rate calculations between the present and the Hawaiian studies may make precise between-study comparisons difficult beyond the fact that the majority of individuals of both populations are marked. The differences in mark rate between encounters may be a result of limited sampling effort and the heterogeneity of data collection. While it may suggest that mark rate varies between groups, the plateau of the discovery rate shows that few new individuals are being encountered in the study area. Results may therefore indicate that subgroups, rather than whole groups, differ in mark rate. Consequently, recorded differences may be due to the bias of observation platforms to focus on approachable subgroups that may not be entirely representative of the whole group that was present but not sampled completely.

3.4.5 Dorsal fin disfigurement

Only 3.3% ($n = 2$) of the photo-identified individuals in the study area are thought to have sustained marks on their dorsal fin as a result of interactions with fisheries. Dorsal fin disfigurement, consistent with cuts made by fishing line, has been linked to interactions between the endangered false killer whales and long-line fisheries in Hawaiian waters and is used as an indicator in the monitoring of such anthropogenic threats. The observed rate of this disfigurement was considerably lower in the present study area compared to Hawaiian waters where rates as high as 13% are reported (Baird and Gorgone 2005, Baird *et al.* 2014a). Additionally, both identified cases in the study area displayed the disfigured dorsal fins when they were first identified. Consequently, no new cases of dorsal fin disfigurement were recorded throughout the course of this study. From a management perspective, this is reassuring, given that false killer whales are considered a ‘problem species’ for pelagic long-line fisheries in numerous locations (*e.g.* Forney and Kobayashi 2008, Ramos-Cartelle and Mejuto 2008, see also Chapter One), resulting in injury and even mortality. Results from this study area would suggest that identified New Zealand false killer whales do not sustain injuries during fisheries interactions at a similar level and/or may interact with fisheries at a considerably lower level than their Hawaiian conspecifics. However, it is worth noting that a beach-cast individual in Hawaii had five fish hooks in its stomach, despite showing no external evidence of fishery interaction (Baird *et al.* 2014a). Consequently, the lack of visible fishery related scarring does not exclude fishery interactions from occurring within the study area. Results should therefore be viewed with caution.

3.4.6 Apparent abundance and apparent survival

The mark-recapture analysis suggests an overall number of 81 marked whales present in the area during the study period, with the superpopulation estimated at 111 individuals. While this estimate is likely conservative, it is comparable to initial estimates of the Hawaiian insular population (123 individuals, Baird *et al.* 2005), which was later adjusted to 151 individuals (Oleson *et al.* 2010). The few available abundance estimates from other regions are noticeably larger but also cover considerably larger areas (1,038 estimated individuals in the northern oceanic Gulf of Mexico (Mullin and Fulling 2004) and 16,668 and 39,800 estimated individuals in the western North Pacific and eastern tropical Pacific, respectively Miyashita 1993, Wade and Gerrodette 1993. Refer also to Table 1.2 in Chapter One).

Results from Goodness of fit tests indicated a ‘trap shy’ effect and the presence of ‘transient’ whales may be occurring. As photo-id itself is unlikely to affect the whales’ behaviour because of the non-invasive nature of the technique, it was concluded that the

heterogeneity observed be likely caused by the uneven sampling effort. Due to the large group sizes and the known dispersal of individuals over extensive areas, it is unlikely that all individuals in every encounter were photographed, although the discovery curve suggests that most individuals occurring in the study area were at least identified once. Missing individuals may therefore have been interpreted as ‘trap shy’. It is also possible that some of the observation platforms may have focused primarily on more ‘approachable’ (or boat-friendly) individuals or those that behaviourally displayed, thereby placing less focus on photographing less interactive individuals. Consequently, heterogeneity may be caused, at least in part, by behavioural responses to observation platforms, with some individuals more boat-tactic than others. This constitutes an ongoing challenge for cetacean research (*e.g.* Mann 1999). In addition, unequal patterns of residency and site fidelity cause heterogeneity in capture probabilities (Tezanos-Pinto 2009, Tezanos-Pinto *et al.* 2013). While a slight decrease in numbers was evident, this was not significant.

The observed transience may also be the result of sampling bias, as it cannot be excluded that individuals were present but simply not captured via photo-id. However, it could also be a result of differing home-range use, as documented in Hawaiian waters (Baird *et al.* 2010). It is worth noting that the rate of new identifications generally decreased over the course of the encounters described herein and that during some encounters no ‘new’ individuals were identified at all. These findings, in addition to the high resighting rate over more than seven years, suggests a relatively small number of individuals with a reasonably high degree of site fidelity, albeit with a strong seasonal component (all sightings recorded between December and May. Refer also to Chapter Two), occurring in the study area. However, due to the heterogeneity of the data sources, these results have to be viewed with caution as they likely represent a minimum estimate. Further data are required to ascertain if these individuals constitute a small local and genetically closed population or form part of a larger, wide ranging metapopulation. Notably, genetic sampling of the false killer whales in Hawaiian waters revealed that individuals with high site fidelity form part of a closed population with a limited home range (Chivers *et al.* 2010, Baird *et al.* 2012).

The apparent survival rate (0.95) is comparable to that estimated for the three social clusters of the Hawaiian insular false killer whale population (0.96, range = 0.95-0.97, Baird *et al.* 2013), which appears to be the only other available survival estimate in the literature. In comparison, survival rate estimates for killer whale populations range from 0.99 and 0.98, respectively for two sympatric populations in Alaskan waters (Matkin *et al.* 2012) to 0.97 in Norway (Kuningas *et al.* 2013) and as low as 0.89 in the Crozet Archipelago (Poncelet *et al.* 2010). Within the study area, the only available survival estimate for a cetacean species is

that for the coastal common bottlenose dolphin population off northeastern New Zealand, which appears to be in decline (0.92, Tezanos-Pinto *et al.* 2013).

The within-year abundance estimates (51-74 marked individuals), together with the leveling off of the discovery curve at 61 *distinctive* or *very distinctive* individuals stands in contrast to the large group sizes of up to 150 individuals reported during some encounters (refer to Chapter Two). Consequently, the presence of a significant number of individuals that were not photo-identified cannot be ruled out and the possibility of some individuals or groups of individuals avoiding the observation vessels has to be considered. However, given that the vast majority of observations described herein comprised of mixed-species groups numbering hundreds of individuals and distributed over large areas, observer error as discussed in Chapter Two, seems the most likely explanation for this discrepancy.

3.5 Conclusion

False killer whales observed in coastal waters of northeastern New Zealand show a relatively high degree of short- and long-term site fidelity despite a strong seasonal peak in occurrence (December - May) and a presumed oceanic distribution. All individuals identified so far are linked by association in one single social network, albeit divided into two apparent clusters, with repeat associations between individuals documented. While further data are required to elucidate questions regarding population size and home range, a small localised and disjunct population as small as 111 individuals appears likely.

These findings highlight the importance of long-term data collection and strongly support the need for dedicated research to be conducted on the species in New Zealand waters. Given the demonstrated high site fidelity, the single social network and low apparent abundance, a reassessment of the current conservation status in New Zealand may also be prudent.

Chapter Four

Interspecific associations



**A mixed-species group of bottlenose dolphins, false killer and pilot whales in the Bay of Islands.
Photo © David Hall.**

4.1 Introduction

Mixed-species groups may occur in a wide range of taxa (*e.g.* fishes, Ward *et al.* 2002, birds, Sridhar *et al.* 2009 and mammals, Stensland *et al.* 2003). Their possible functions have long since intrigued scholars (*e.g.* Morse 1977, Stensland *et al.* 2003, Cords and Würsig 2014), likely because the causative factors behind the associations can be difficult to identify due to the considerable range of potential costs and benefits for each respective species (Cords and Würsig 2014). The investigation of possible drivers of mixed-species groups (also known by the synonyms of heterospecific groups, Heymann and Buchanan-Smith 2000) follows many of the theories devised for group living in general (*e.g.* Norris and Schilt 1988). Groups are generally viewed in terms of the potential advantages that they afford their participants, compared to non-joiners, with costs and benefits usually categorised as either direct or indirect to individual fitness (Acevedo-Gutiérrez 2008). Furthermore, groups are generally classed by their degree of sophistication and duration, ranging from ephemeral aggregations around a common attractor, such as prey (Cords and Würsig 2014), to complex life-long teams such as is evident in many insect societies (Robinson 1992). Heterospecific and homospecific groups may differ in a number of aspects; in heterospecific groups the role of kin selection is generally absent or strongly reduced, and the flexibility of maintaining and/or terminating the association is potentially greater as is the likelihood of diverging skill sets. The primary benefits suggested for most mixed-species groups are predation evasion and/or improved foraging, with social factors possibly also playing a role in some associations (Norris and Schilt 1988). However, the nature of heterospecific groups can range from cooperative mutualism (*e.g.* foraging birds, Hino 1998) to social parasitism (*e.g.* ants, Buschinger 2009), with costs and benefits often unequally divided between participating species (Cords and Würsig 2014).

While numerous accounts of mixed-species groups of cetaceans exist in the literature (*e.g.* Baraff and Asmutis-Silva 1998, Frantzis and Herzing 2002, Jefferson *et al.* 2006), few studies have attempted to actually test for possible drivers. This is likely due to the logistical challenges of studying cetacea in the field. The few exceptions that exist have found plausible evidence for anti-predatory strategies (*e.g.* Kiszka *et al.* 2011), and to a lesser extent improved foraging and/or social factors (Stensland *et al.* 2003, Quéroil *et al.* 2008). Mixed-species groups including false killer whales (*Pseudorca crassidens*) are considered relatively common (Baird 2008), with the species known to associate with a number of other delphinid species (*e.g.* Leatherwood *et al.* 1989), in particular with the common bottlenose dolphin (*Tursiops truncatus*, hereafter referred to as bottlenose dolphin) (*e.g.* Kasuya 1985, Flores *et al.* 2003, Best and Reeb 2010). Nonetheless, investigations of the possible

functions of these groups are extremely scarce, with a literature search revealing only one study from Japanese waters, suggesting joint foraging as a possible driver based solely on the apparent seasonal aspect of the association (Kasuya 1986). The frequency of the documented interspecific associations, both, in the literature and in this study, set against the almost complete paucity of studies aiming to identify their possible drivers, warrants further investigation. This chapter presents the extent and complexity of the frequently observed associations between false killer whales and bottlenose dolphins off northeastern New Zealand, using photo-identification (photo-id) and spatial and temporal parameters to elucidate their possible function/s (refer also to Visser *et al.* 2010, Appendix B, Zaeschmar *et al.* 2013, Appendix C and Zaeschmar *et al.* 2014, Appendix D). Studies of interspecific associations between cetacean species from the literature are reviewed and discussed in the context of general group theory and comparisons to other taxa, with the aim to produce plausible hypotheses.

4.2 Methods

4.2.1 Study area

All sightings were collected in five locations along an approximate 650 kilometer (km) stretch of the northeastern coast of New Zealand, between February 1995 and March 2012, with the majority of encounters recorded in the Bay of Islands (BOI). Additional records were collated from three locations south of the BOI and one location north of the BOI, where whale-watch operations and/or cetacean research was carried out: The Three Kings Islands (TKI, 2008), the Poor Knights Islands (PKI, 2010), the Hauraki Gulf (HG, 2011), the Bay of Plenty (BOP, 2012). A more detailed description of the study area and the five particular locations within it, is provided in the methods section of Chapter Two.

4.2.2 Survey platforms and methods

Of the 47 sightings analysed herein, the majority of records (53.2%, $n = 29$) were collected from the *Tutunui*, a commercial whale-watching vessel staffed by experienced marine mammal observers that operates year round in BOI. Additional records were collected opportunistically from wildlife/marine tour boats and via dedicated cetacean research vessels operating in the five study locations (Refer to Chapter Two, Figure 2.4).

All tour boats followed a similar, asystematic survey methodology, which was dictated by factors such as prevailing weather conditions, but also suspected areas of likely cetacean

occurrence and/or sighting reports from other vessels. The research vessels conducted dedicated cetacean surveys when visibility was $> 1\text{km}$ and Beaufort sea-state was ≤ 3 . A continuous scanning methodology (e.g. Mann 1999) was employed, with trained observers using both naked eye and binoculars. A more detailed description of the survey platforms and methods are provided in the methods section of Chapter Two.

4.2.3 Definition of mixed-species group

Given the frequency of association between false killer whales and bottlenose dolphins as described in Chapter Two, for the purpose of this study, the term mixed-species group refers to associations between these two species only. Following Shane (1990), a mixed-species group was defined as any number of individuals of one species observed in apparent association with the other species and generally moving in the same direction and engaged in similar behaviour.

4.2.4 Definitions of coastal and oceanic bottlenose dolphins

Two forms (coastal and oceanic) of the bottlenose dolphin frequent New Zealand waters (Baker *et al.* 2010). The oceanic form is distinguishable based on gross morphology (Visser *et al.* 2010); they are comparatively more robust and typically exhibit wounds and scars, presumed to be inflicted by the cookie cutter shark (*Isistius* spp.) (Constantine 2002, Dwyer and Visser 2011). In contrast, the New Zealand coastal form does not usually exhibit cookie cutter shark scarring (Constantine 2002). Little is known about the oceanic form. They are occasionally sighted in deeper waters of the study area during late summer months but are not considered part of the coastal population (Constantine 2002, Tezanos-Pinto 2009) and no interactions between the two forms have been reported from the study area (Tezanos-Pinto 2009). Consequently, the presence or absence of cookie cutter scars was determined from photographs of the respective encounters to ascertain if the bottlenose dolphins observed in association with false killer whales were of the oceanic or the coastal form. Individuals with visible fresh or healed presumed cookie cutter bite marks were counted and totals were compared to the number of individuals without such marks to produce a minimum proportion of individuals with cookie cutter scarring.

4.2.5 Seasonality of mixed-species associations

The possible seasonal aspects of interspecific associations between the two species were assessed only from the records of *Tutunui*, which ran continuous trips up to twice daily in BOI between 1995 and 2007. Records from Great Sights' *Tutunui* were chosen over those from the other platforms in this study as they were verifiable, of significant number and

were collected during continuous year-round operation. Following Wiseman *et al.* (2011), a monthly index of false killer whale encounters was determined using a trip encounter rate (TER), which was calculated from the number of trips during which the species was encountered in proportion to the total number of trips undertaken that month. To increase the power of analysis, sightings were further pooled into warm (December – May) and cold (June – November) seasons. A Kolmogorov-Smirnov test was applied using Minitab 17⁹ to test for normality. As data were not normally distributed (Kolmogorov-Smirnov: $Z = 0.481$, $SD = 0.013$, $p = 0.010$), the non-parametric Kruskal-Wallis test was applied.

4.2.6 Photo-identification

Standard photo-id methods (Würsig and Jefferson 1990, Baird *et al.* 2008) were applied to identify individuals by the distinctive permanent marks (nicks or notches) on the leading and trailing edges of the dorsal fins. Individuals, as well as images, were graded according to the likelihood of successful recapture and matching as per Baird *et al.* 2008. The quality of each image was assessed by its focus, contrast and the angle of the fin relative to the frame and subsequently graded on a scale of 1 to 4, with 1 being *excellent*, 2 being *good*, 3 being *fair* and 4 being *poor* as per Baird *et al.* 2008. Likewise, the distinctiveness of each dorsal fin was additionally graded on a scale of 1 to 4, with 1 being *very distinctive*, 2 being *distinctive*, 3 being *slightly distinctive* and 4 being *not distinctive* as per Baird *et al.* 2008 (refer also to Chapter Three, Tables 3.2 and 3.3). Only *very distinctive* and *distinctive* individuals and images of *excellent* or *good* quality were included in the analysis (Baird *et al.* 2008). Successful photo-ids were entered into the New Zealand False Killer Whale Identification Catalogue (NZFKWC, Zaeschmar unpubl. data), and New Zealand Oceanic Bottlenose Dolphin Identification Catalogue (NZOBDC, Zaeschmar unpubl. data), respectively. A more detailed description of the photo-id methods used can be found in Chapter Three.

4.2.7 Social network analysis

A social network diagram of false killer whales and bottlenose dolphins observed in association in the study area was produced using the program *Netdraw 2.123*¹⁰. A spring embedded layout was chosen, placing more connected nodes at the centre of the diagram, while those with fewer connections were placed around the periphery.

⁹ Minitab Inc. (2010). URL: www.minitab.com.

¹⁰ Borgatti, S.P., 2002. NetDraw Software for Network Visualization. Analytic Technologies: Lexington, KY Available from analytictech.com/Netdraw

4.3 Results

4.3.1 Interspecific association with bottlenose dolphins

Of the 47 false killer whale encounters in the study area, the majority (91.5%, $n = 43$) were recorded in association with bottlenose dolphins (Figure 4.1). Group sizes for bottlenose dolphins were estimated during 32 encounters and ranged from five to *ca.* 250 individuals ($\bar{x} = 62.8$, $SD = 42.8$). Spatial separation/dispersion of focal groups varied, ranging from all individuals being in close proximity (total area estimated at $< 1,000 \text{ m}^2$, Figure 4.2) to wide distribution of individuals and subgroups within the whole mixed-species group (total area estimated at $> 5 \text{ km}^2$). During all encounters, the two species were generally behaving as a single group, swimming within a body length of the other species (Figure 4.3). However, clear segregation into conspecific subgroups within the focal group was also reported during two encounters (PKI 2010 and HG 2011, Figure 4.4).

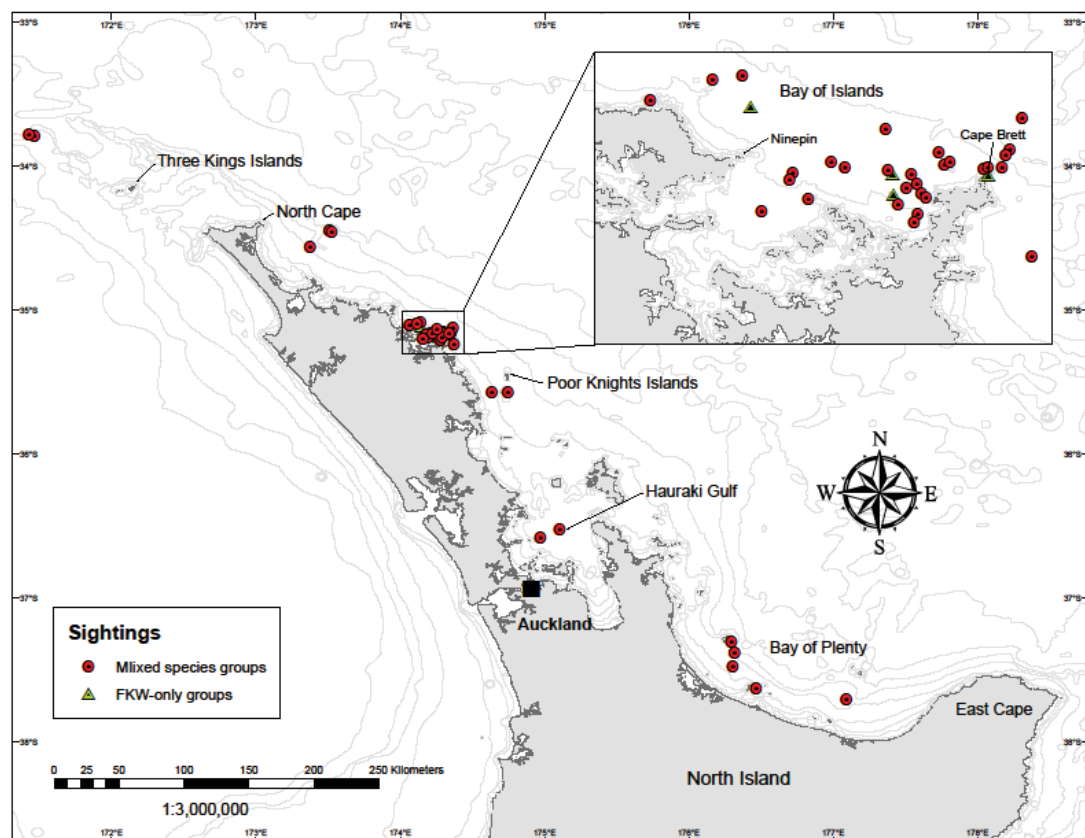


Figure 4.1. Map showing the 47 locations of opportunistic false killer whale sightings off northeastern New Zealand between 1995 and 2012, including the 43 encounters where associations with bottlenose dolphins were observed. Red circles indicate mixed-species groups of false killer whales and bottlenose dolphins. Green triangles indicate groups of false killer whales only.



Figure 4.2. A mixed-species group of false killer whales and bottlenose dolphins. Bay of Islands, New Zealand, January 2007. Photo © David Hall.



Figure 4.3. Close interspecific associations between a false killer whale and a bottlenose dolphin off northeastern New Zealand. Photo © Mazdak Radjainia.



Figure 4.4. A mixed-species group of false killer whales and bottlenose dolphins showing clear segregation into conspecific subgroups, with false killer whales in the top right hand quarter of the image and bottlenose dolphins at the lower left quarter. Poor Knights Islands, New Zealand, April 2010 Photo © Richard Robinson.

4.3.2 Photo-identification

Cookie cutter shark scars were reported during all 43 encounters. These could be quantified from 1,230 photographs of *excellent* or *good* quality, taken during 15 encounters (BOI = 7; BOP = 3; TKI = 2; HG = 2; PKI = 1), and showed that the majority of photographed individuals ($\bar{x} = 74.7\%$, range 69.7-80.0%, SD = 3.0) exhibited visible cookie cutter scars (Figure 4.5, Table 4.1). Based on this consistently high rate of scarring and the robust morphology observed, the bottlenose dolphins encountered in association with false killer whales were presumed to be of the oceanic form. This assumption is further supported by the lack of observed interactions between the coastal and the oceanic form in the study area despite ongoing long-term research of the coastal form (*e.g.* Constantine *et al.* 2004, Tezanos-Pinto 2013).

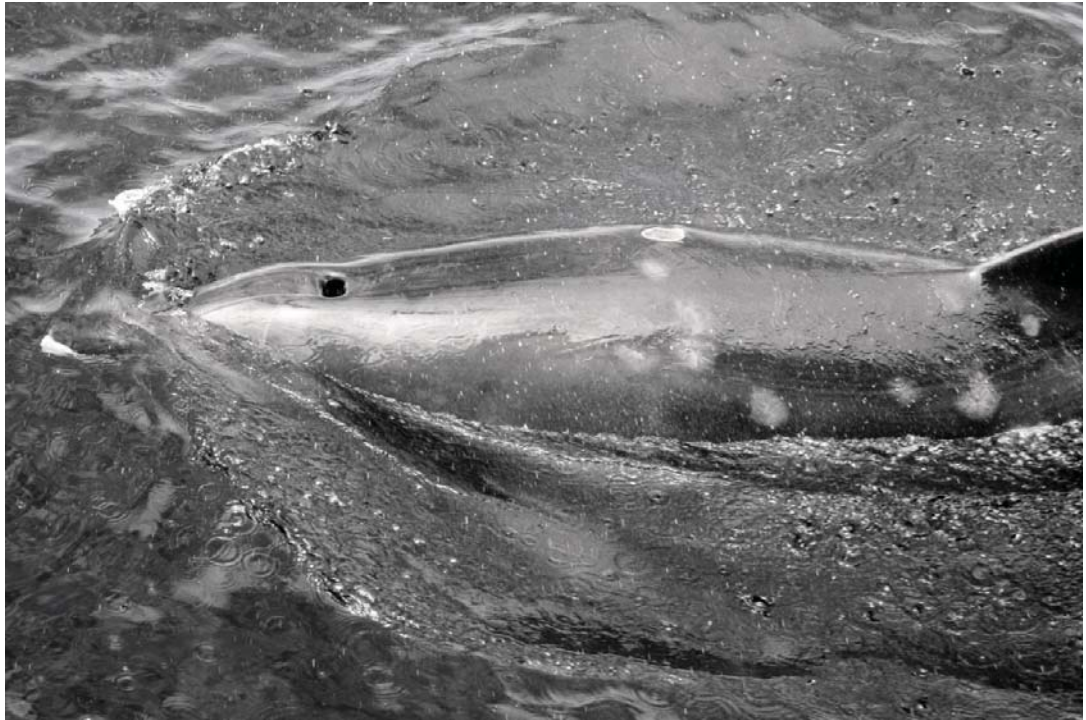


Figure 4.5. A presumed oceanic common bottlenose dolphin showing the characteristic fresh and healed scars likely inflicted by the cookie cutter shark (*Isistius* spp.). Bay of Islands, New Zealand 2009. Photo © Lara Kay/Explore Images.

Table 4.1. Rates of visible cookie cutter shark bite scars observed on presumed oceanic bottlenose dolphins seen in association with false killer whales during 14 encounters off northeastern New Zealand between 2005 and 2012. Area codes are Bay of Islands (BOI), Bay of Plenty (BOP), Hauraki Gulf (HG), Poor Knights Islands (PKI) and the Three Kings Islands (TKI). Observations were made from tour boats (T) and/or research vessels (R). All observations were of mixed-species groups of false killer whales and bottlenose dolphins.

Date	Area	Vessel type	Total photos taken	Total dolphins with scars	Total dolphins without scars	Percentage of dolphins with visible scars
2005-Feb-11	BOI	R	12	8	3	75.0
2007-Jan-11	BOI	R	144	113	31	78.5
2007-Apr-03	BOI	R	64	47	17	73.4
2007-Dec-28	BOI	R	175	120	44	69.7
2008-May-16	TKI	T	11	8	3	72.7
2008-May-17	TKI	T	25	19	6	76.0
2009-Feb-25	BOP	T	32	24	8	75.0
2009-Dec-21	BOI	R	314	242	72	77.1
2010-Mar-20	BOI	T	21	15	6	71.4
2010-Apr-20	PKI	T	8	6	2	75.0
2011-Jan-20	HG	T	176	127	49	72.2
2011-Jan-25	HG	R	90	72	18	80.0
2012-Jan-18	BOP	R	102	73	29	71.6
2012-Feb-09	BOP	R	92	72	20	78.3
Mean			90.4	67.6	22.0	74.7

4.3.3 Seasonality and range

The seasonality of the occurrence of mixed-species groups followed a very similar trend to that described for false killer whale seasonality in Chapter Two, given that the vast majority of encounters from the study area (TKI to BOP, 91.5%, $n = 43$) and also from *Tutunui* in BOI (86.2%, $n = 25$) comprised mixed-species groups. The two species were observed in association during all austral summer and autumn months (December – May, Figure 4.6). TER analysis from BOI records indicated abnormal distribution (Kolmogorov-Smirnov: $Z = 0.481$, $SD = 0.013$, $p = 0.010$). No significant differences between seasons (Kruskal-Wallis: $Z = 7.61$, $d.f. = 1$, $p = 0.006$), months (Kruskal-Wallis: $Z = 4.71X$, $d.f. = 11$, $p = 0.952$) or years (Kruskal-Wallis: $Z = 3.621$, $d.f. = 12$, $p = 0.975$) was detected. Seasonality for sightings of mixed-species groups from the other locations in the study area were BOP: January, ($n = 1$), February, ($n = 2$) and March, ($n = 2$), HG: (January, $n = 2$), PKI: February, ($n = 1$) and April, ($n = 1$) and TKI: April ($n = 3$) and May, ($n = 2$).

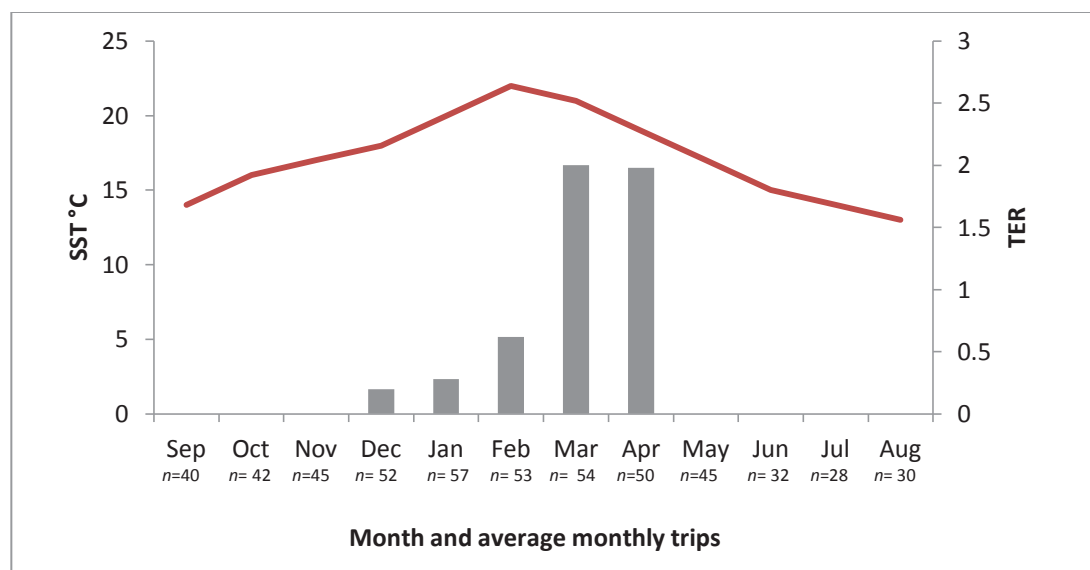


Figure 4.6. Seasonality of 43 mixed-species groups of false killer whales and bottlenose dolphins assessed from trip encounter rates (TER), calculated from the records of the Tutunui in the Bay of Islands, New Zealand between 1995 and 2007. Sea surface temperature (SST) is shown as a red line. The seasonality of false killer whale only groups is not shown due to the small sample size ($n = 4$).

4.3.4 Resightings of individuals

Between 2005 and 2012, *excellent* or *good* images of dorsal fins of bottlenose dolphins observed in association with false killer whales were obtained during nine encounters (BOI = 4; BOP = 2; HG = 2; TKI = 1, Figure 4.7). Using only *excellent* and *good* quality photographs, 242 identifications of 163 individuals were made ($\bar{x} = 1.48$, $SD = 0.71$), 8.6%

($n = 14$) of which were classified as only *slightly distinctive* and not included in the analysis. This resulted in 217 identifications of 149 *very distinctive* or *distinctive* individuals ($\bar{x} = 1.45$, $SD = 0.77$). Of these, 34.2% ($n = 51$) were resighted, with 10.1% ($n = 15$) observed on three or more occasions and two individuals (1.3%) on four occasions.

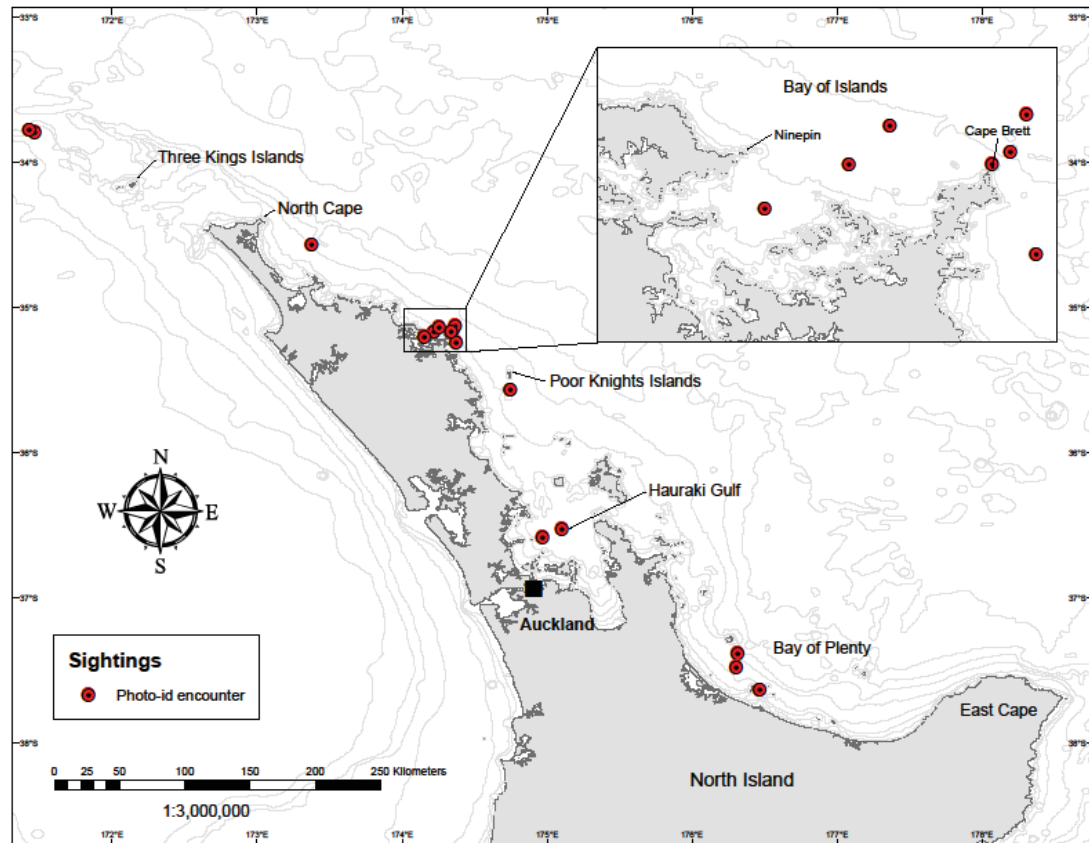


Figure 4.7. Study area showing the encounter locations ($n = 9$), where photo-identification of mixed-species groups of false killer whales and bottlenose dolphins was carried out off northeastern New Zealand between 2007 and 2012.

Additionally, 28.2% ($n = 42$) were encountered in more than one year (Figure 4.8) and 23.4% ($n = 35$) were observed in more than one of the five sighting locations within the study area (Figure 4.9, Table 4.2). The longest time between first identification of an individual and its most recent resighting was 1,832 days (*ca.* 5 years) documented for two individuals (Figure 4.10). Distances between sighting locations ranged from < 1 km (BOI, 1,074 days, $n = 14$) to *ca.* 650 km (TKI – BOP, 284 days, $n = 1$).

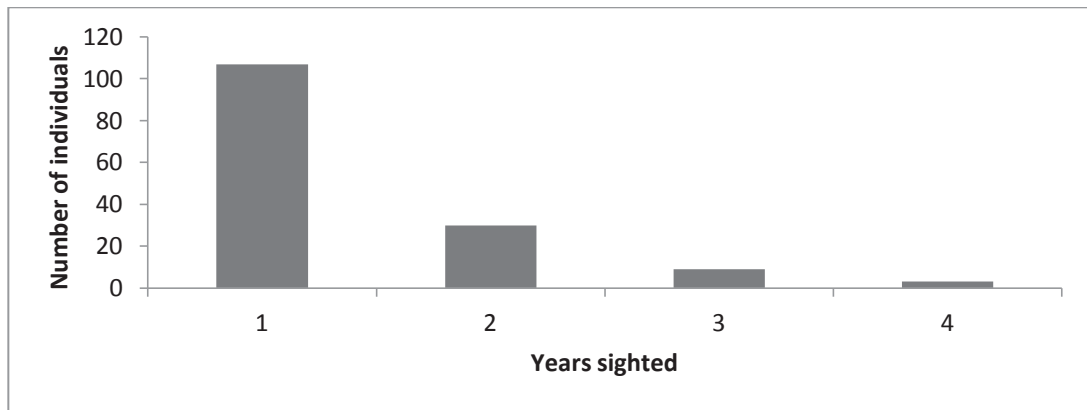


Figure 4.8. Number of different years in which identified individual bottlenose dolphins were observed in association with false killer whales off northeastern New Zealand between 2007 and 2012.

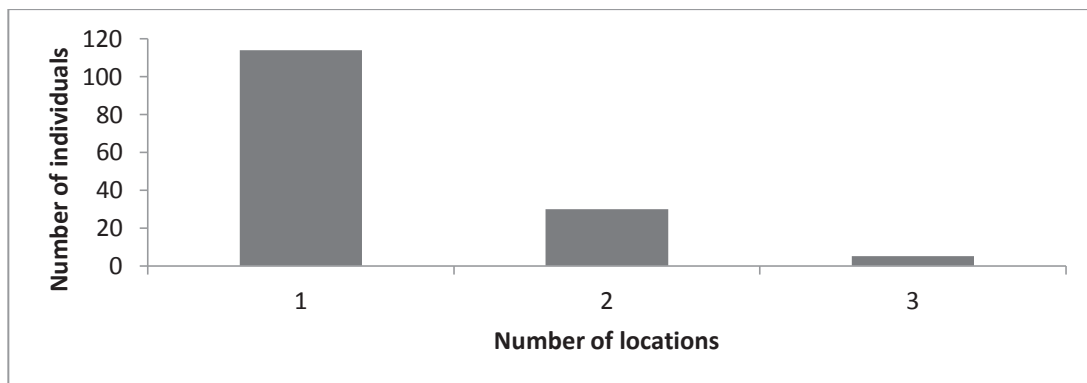


Figure 4.9. Number of different locations within the study area where individual bottlenose dolphins were sighted in association with false killer whales off northeastern New Zealand, between 2007 and 2012.

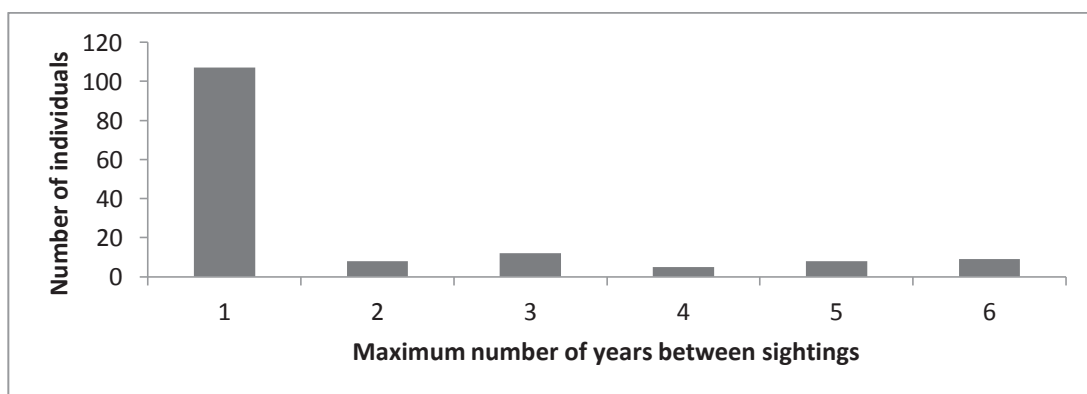


Figure 4.10. Maximum number of years between sightings of individual bottlenose dolphins observed in association with false killer whales off northeastern New Zealand, between 2007 and 2012.

Table 4.2. Photo-identified *very distinctive* and *distinctive* bottlenose dolphins observed on more than one occasion in mixed-species groups with false killer whales off northeastern New Zealand: Three Kings Islands (TKI), Bay of Islands (BOI), Hauraki Gulf (HG) and Bay of Plenty (BOP), showing movements between the different encounter locations and associations between individuals, 2007-2012.

Dolphin ID	Encounter date								
	17 May 2008	11 Jan 2007	03 Apr 2007	28 Dec 2007	21 Dec 2009	20 Jan 2011	25 Jan 2011	18 Jan 2012	09 Feb 2012
	Estimated group size								
	60	50	50	100	80	150	60	200	150
	Encounter location								
	TKI			BOI		HG		BOP	
NZ-OTi-002	Δ		Δ						
NZ-OTi-009	Δ				Δ				Δ
NZ-OTi-010	Δ				Δ		Δ		
NZ-OTi-013	Δ				Δ				
NZ-OTi-015	Δ			Δ	Δ				
NZ-OTi-016	Δ			Δ					
NZ-OTi-017	Δ							Δ	
NZ-OTi-018	Δ						Δ		
NZ-OTi-023	Δ		Δ	Δ				Δ	
NZ-OTi-024	Δ			Δ	Δ			Δ	
NZ-OTi-025	Δ		Δ						
NZ-OTi-028	Δ				Δ				
NZ-OTi-031	Δ				Δ				
NZ-OTi-035	Δ				Δ			Δ	
NZ-OTi-036	Δ					Δ			Δ
NZ-OTi-037				Δ	Δ				
NZ-OTi-038	Δ				Δ				
NZ-OTi-039	Δ				Δ		Δ		
NZ-OTi-042	Δ		Δ						
NZ-OTi-044			Δ	Δ					
NZ-OTi-046			Δ	Δ	Δ				
NZ-OTi-047				Δ	Δ				
NZ-OTi-048			Δ	Δ	Δ			Δ	
NZ-OTi-049					Δ			Δ	
NZ-OTi-054				Δ	Δ				
NZ-OTi-057				Δ	Δ				
NZ-OTi-059				Δ	Δ				
NZ-OTi-064				Δ	Δ				
NZ-OTi-065				Δ			Δ		
NZ-OTi-067				Δ	Δ				
NZ-OTi-068					Δ	Δ	Δ		
NZ-OTi-070				Δ			Δ		
NZ-OTi-075					Δ	Δ		Δ	
NZ-OTi-077					Δ	Δ			
NZ-OTi-081					Δ			Δ	
NZ-OTi-084					Δ			Δ	
NZ-OTi-085					Δ			Δ	
NZ-OTi-091					Δ		Δ	Δ	
NZ-OTi-114						Δ			Δ
NZ-OTi-115			Δ			Δ			
NZ-OTi-117						Δ	Δ		
NZ-OTi-119						Δ		Δ	
NZ-OTi-121						Δ		Δ	Δ
NZ-OTi-122						Δ	Δ	Δ	
NZ-OTi-124							Δ	Δ	
NZ-OTi-131							Δ	Δ	
NZ-OTi-133							Δ	Δ	
NZ-OTi-139								Δ	Δ
NZ-OTi-144								Δ	Δ
NZ-OTi-150	Δ							Δ	Δ
NZ-OTi-152								Δ	Δ

The number of dolphins identified in each encounter ranged from 1-54 individuals ($\bar{x} = 24.1$, $SD = 16.5$, $n = 149$). Every encounter, subsequent to the initial encounter, included previously photo-identified individuals. However, the proportion of resighted individuals was always smaller than that of previously unidentified individuals (Figure 4.11).

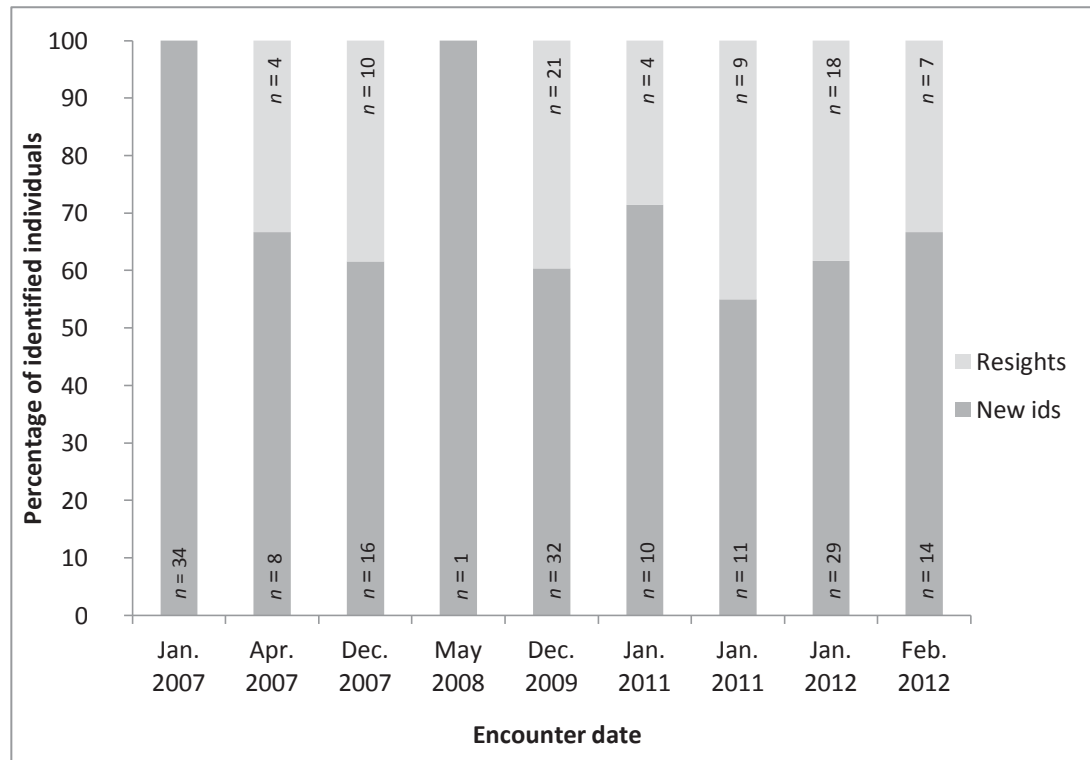


Figure 4.11. Percentages of newly (New ids) and previously (Resights) photo-identified bottlenose dolphins per encounter off northeastern New Zealand between 2007 and 2012.

The highest number of resightings between any two encounters was 11 individuals. At least 34.2% of bottlenose dolphins identified ($n = 51$) had repeat associations with a minimum of 67.2% ($n = 41$) of the 61 false killer whales included in the photo-id analysis. Of these, 29.4% of bottlenose dolphins ($n = 15$) and 87.8% of false killer whales ($n = 36$) were re-encountered together on three or more occasions and during more than one year. In addition, 23.5% of dolphins ($n = 12$) and 66.7% of whales ($n = 28$) were observed together on three or more occasions as well as in different years and locations (Table 4.3).

Table 4.3. Repeat associations between individual dolphins and false killer whales off northeastern New Zealand: Bay of Islands (BOI), Hauraki Gulf (HG) and Bay of Plenty (BOP) between 2007 and 2012. Only *very distinctive* and *distinctive* individuals observed together on three or more occasions and during more than one year are shown.

Dolphin ID	Whale ID	Encounter date							
		11 Jan 2007	03 Apr 2007	28 Dec 2007	21 Dec 2009	20 Jan 2011	25 Jan 2011	18 Jan 2012	09 Feb 2012
		Encounter location							
NZ-OTt-	NZ-Pc-	BOI			HG		BOP		
009	002, 004, 011, 013, 015, 016, 018, 019, 020, 031	◇			◇			◇	
010, 039	003, 004, 013, 015, 019, 021, 024, 026, 028	◇			◇		◇		
015	001, 002, 003, 004, 005, 007, 010, 011, 012, 013, 015, 016, 017, 018, 019, 020, 022, 024, 025, 028, 030, 032, 033	◇		◇	◇				
023	001, 002, 003, 005, 013, 015, 016, 018, 019, 020, 022, 024	◇		◇				◇	
023	008	◇	◇	◇					
024	001, 002, 003, 005, 013, 015, 016, 018, 019, 020, 022, 024	◇		◇	◇			◇	
024	027, 031	◇			◇			◇	
035	002, 004, 011, 013, 015, 016, 018, 019, 019, 020, 022, 024, 027, 031	◇			◇			◇	
036	004, 013, 016, 018, 020, 031	◇				◇		◇	
046, 048	007		◇	◇	◇				
068	004, 013, 024, 036, 041, 061, 062				◇	◇	◇		
075	001, 005, 013, 016, 018, 020, 024, 031, 061, 062, 064, 066				◇	◇		◇	
091	003, 013, 015, 019, 022, 024, 061, 062				◇		◇	◇	
121	013, 016, 018, 020, 031, 062					◇		◇	
122	013, 024, 061, 062					◇	◇	◇	

4.3.5 Social network

Of the 61 *very distinctive* and *distinctive* false killer whales and 145 *very distinctive* and *distinctive* bottlenose dolphins photo-identified in the study area, all were linked by association in one large social network. In addition to the two false killer whale social clusters identified in Chapter Three, several clusters of bottlenose dolphins were apparent. Additionally, one false killer whale social cluster appears to have fewer associations with bottlenose dolphins than the other (Figure 4.12).

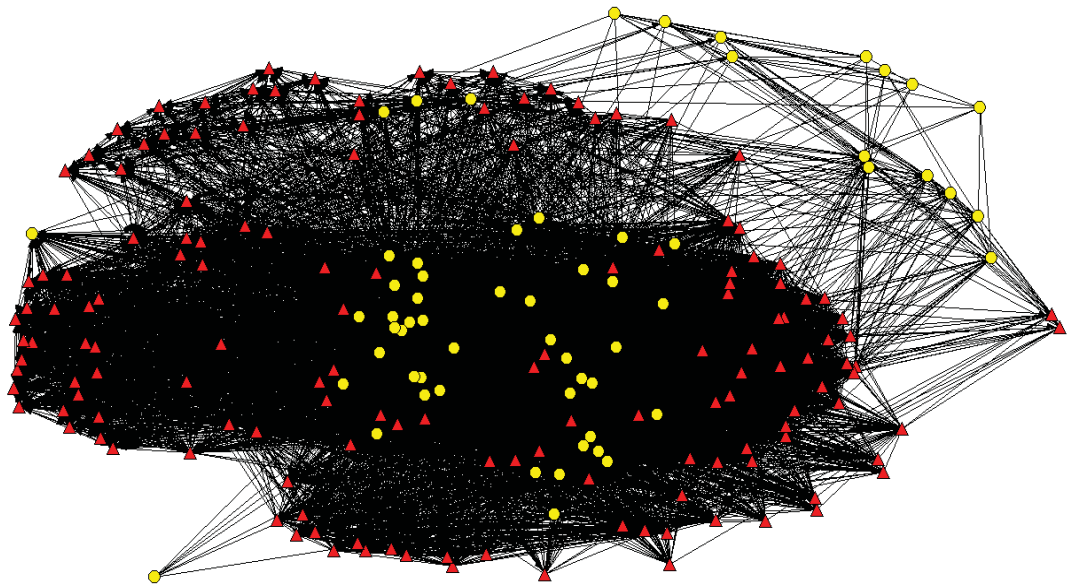


Figure 4.12. Social network diagram of 61 *very distinctive* or *distinctive* false killer whales and 149 *very distinctive* or *distinctive* bottlenose dolphins photo-identified off northeastern New Zealand during 9 encounters between 2007 and 2012 using a spring embedded layout. Individual false killer whales are represented by yellow circles and bottlenose dolphins are represented by red triangles. Only individuals with good or excellent quality photos are included.

4.3.6 Interspecific associations with other cetacean species

In addition to the observed associations between false killer whales and bottlenose dolphins, a third species was present during 8.5% ($n = 4$) of encounters (killer whale, *Orcinus orca*, $n = 2$, long-finned pilot whale, *Globicephala melas*, $n = 2$).

4.3.6.1 Killer whales

On the first of the two observed interactions in 2005, a mixed-species group of *ca.* 60 false killer whales and 80 bottlenose dolphins, which had been travelling steadily in the same

direction, was observed slowing abruptly upon approaching within *ca.* 500 meters (m) of a group of three adult killer whales. After a *ca.* 30 second period of non-directional movement, the mixed-species group made a *ca.* 135° directional change away from the killer whales and started moving at an accelerated speed than previously observed. The group of three killer whales remained in the vicinity and did not show any obvious behavioural or course changes.

During the second encounter in 2010 (described in detail in Visser *et al.* 2010, Appendix B), a mixed-species group of *ca.* 50 false killer whales and five bottlenose dolphins was observed being pursued by a group of eight killer whales. The mixed-species group was swimming in a tight formation and after a brief period of non-directional movement, started to accelerate away from the killer whales but in the direction of the coast *ca.* 2 km away. Subsequently, at least one killer whale approached the mixed-species group and rammed an adult false killer whale from below, with a total of five false killer whales rammed and pushed out of the water in four separate events within just over a minute (min). Predation on one false killer whale calf was observed (Figure 4.13).

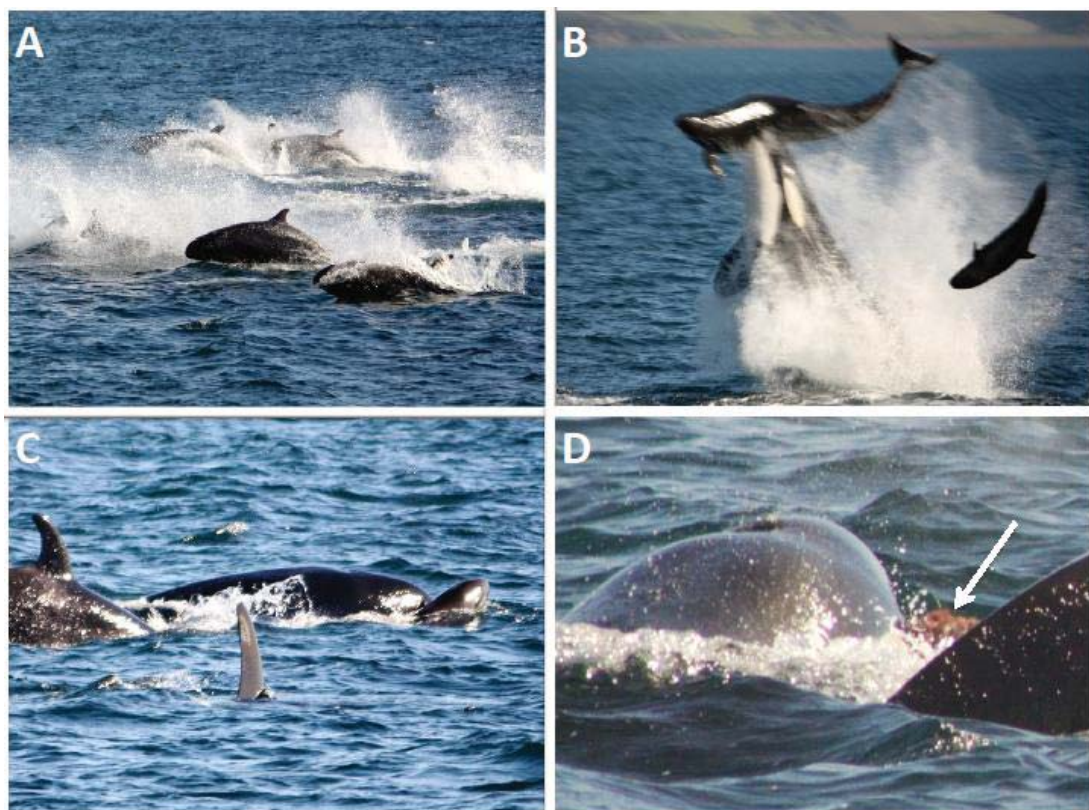


Figure 4.13. Sequence illustrating killer whale predation on false killer whales in the Bay of Islands, New Zealand in March 2010. (A) High speed pursuit. (B) Ramming of an adult false killer whale clear of the water. (C) Herding of a false killer whale calf. (D) Feeding on the carcass of the calf (or parts thereof). Photos © Explore Images.

4.3.6.2 Long-finned pilot whales

During two encounters (BOI 2007, TKI 2008), mixed-species groups were observed in close proximity to long-finned pilot whales. During the BOI encounter, a group of *ca.* 25 long-finned pilot whales approached a mixed-species group of *ca.* 50 false killer whale and 50 bottlenose dolphins, which had been engaged in non-directional travel in the area for *ca.* 45 min. When the long-finned pilot whales were within *ca.* 500 meters of the mixed-species group, the latter approached the long-finned pilot whales rapidly and close associations of the three species ensued, with individuals at times less than a body length away from heterospecifics (Figure 4.14). After *ca.* 20 min, the long-finned pilot whales departed the mixed-species group, travelling slowly in the reciprocal direction they had approached from, while the mixed-species group remained in the area.

During the TKI 2008 encounter, a mixed-species group of *ca.* 90 bottlenose dolphins and 30 false killer whales was observed in association with a group of *ca.* 20 long-finned pilot whales. The groups were travelling slowly in a consistent direction and remained closely associated at < 5 body lengths apart. The pilot whales remained in a tight group within the larger group, showing no signs of mixing with the other species. The encounter was terminated after *ca.* 20 min, due to inclement weather conditions, with all three species continuing to travel slowly in the same direction.

A total of 11 pilot whales (BOI, $n = 8$, TKI, $n = 3$) were photo-identified during these encounters, with no individuals sighted during more than one occasion. Additionally, 41 false killer whales (BOI, $n = 37$, TKI, $n = 4$) and 43 bottlenose dolphins (BOI, $n = 41$, TKI, $n = 2$) were identified during both encounters. Of these, all four false killer whales present in TKI but none of the two bottlenose dolphins were photographed during both encounters. However, both bottlenose dolphins have been observed in mixed-species groups during other encounters in the course of this study, with one individual observed in BOI in 2009 and the other in HG in 2011 and BOP in 2012.

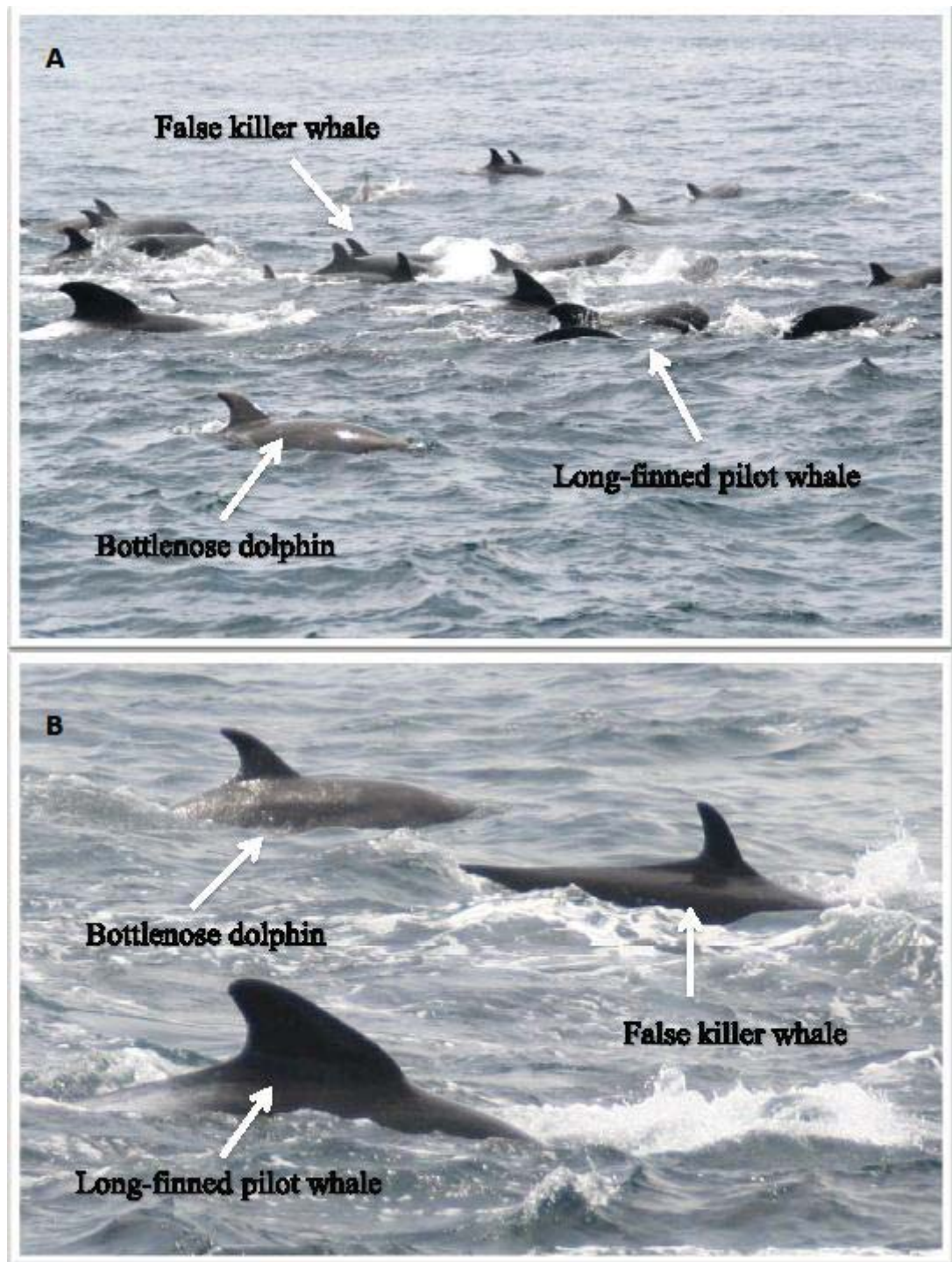


Figure 4.14. Close interspecific associations between false killer whales, pilot whales and bottlenose dolphins. Bay of Islands, New Zealand 2007. Photo © Alex Black.

4.4 Discussion

The extremely high frequency of associations with bottlenose dolphins in the study area, including repeat associations between individual false killer whales and bottlenose dolphins spanning more than five years and up to 650 km, suggests that such associations are not random. In contrast to the more stable group structure identified for false killer whales in the study area (Refer to Chapter Three), bottlenose dolphin groups appeared to be considerably more variable, both in terms of size and membership. These observations are consistent with the respective social structures reported for each species. While the fission-fusion societies of bottlenose dolphins are extensively documented (*e.g.* Shane and Wells 1986, Constantine *et al.* 2004, Krützen *et al.* 2004), the few existing studies of false killer whale social structure suggest a more rigid group structure (*e.g.* Connor *et al.* 1998, Baird *et al.* 2010, refer also to Chapter Three). Consistent with these traits, the mixed-species groups observed in the present study area comprise relatively stable groups of false killer whales in almost constant association with more fluid groups of bottlenose dolphins.

Based on the extensive cookie cutter shark bites, the bottlenose dolphins observed in these mixed-species groups are unlikely to be part of the coastal populations that show high site-fidelity in the study area. Furthermore, preliminary photo-id comparisons between the individuals identified in this study and those from coastal catalogues within the study area have not yielded any matches so far (Zaeschar *et al.* in prep.). Like false killer whales, oceanic bottlenose dolphins are rarely encountered in the study area and little is known about this ecotype in New Zealand waters (Baker *et al.* 2010). Most records from the study area report the species in association with false killer and/or long-finned pilot whales, although homospecific groups are also observed occasionally (Zaeschar unpubl. data). The high rate of discovery of previously unidentified individuals, in particular in contrast to that of false killer whales, suggests that only a fraction of the population of oceanic bottlenose dolphins has been identified to date.

Due to the almost permanent nature of the association, no spatial or temporal trends could be identified beyond the strong seasonal occurrence of false killer whales across the entire study area between December and May, as documented in Chapter Three. Oceanic bottlenose dolphins appear to be absent from the study area in winter (Zaeschar unpubl. data). Seasonal occurrence of oceanic bottlenose dolphins in coastal waters has been indicated in Japanese waters, where the species is associated with seasonally occurring fish stocks (Kasuya 1985). It is therefore possible that the oceanographic processes described in Chapter Two may also affect the presence of oceanic bottlenose dolphins in the study area.

Social network analysis indicates that all identified members of observed mixed-species groups are linked by association in one large social network, albeit with some apparent clustering. As has been highlighted in Chapter Three, clustering may be the result of incomplete sampling, which is also likely the cause of the apparent differences in associations with bottlenose dolphins between the two false killer whale social clusters.

The extremely high frequency of association between the two species raises questions as to its possible function/s. False killer whales are known to associate with other delphinids and have been observed in close, non-aggressive association with a number of cetacean species (*e.g.* Leatherwood *et al.* 1989, Odell and McClune 1999), in particular the bottlenose dolphin (Mizue and Yoshida, 1961, Tsutsumi *et al.* 1961, Zhou *et al.* 1982, Leatherwood *et al.* 1989, Scott and Chivers 1990, Flores *et al.* 2003, Anderson 2005, Maze-Foley and Mullin 2006, Baird *et al.* 2008, Best and Reeb 2010). However, although bottlenose dolphin and false killer whale associations are reported previously in the literature and are considered relatively common (Reeves *et al.* 2002), very few studies have so far attempted to elucidate the possible extent and/or function of these associations. Tsutsumi *et al.* (1961) propose the availability of food resources as a possible driver behind false killer whale and bottlenose dolphin associations in Japanese waters and further state that associations appear to be seasonal in the region. However, the apparent long-term association between individual false killer whales and bottlenose dolphins, as documented in this study, has not been previously reported from other regions (although it seems likely that it does occur elsewhere).

While mixed-species associations of a wide range of cetacean species are regularly observed (*e.g.* Jefferson *et al.* 2006, May-Collado 2010, Fulling *et al.* 2011, Smultea *et al.* 2014), most appear to be short-lived (*e.g.* Herzing and Johnson 1997, Migura and Meadows 2002, Psarakos *et al.* 2003). Only a few records of long-term interspecific associations between individuals of any two species of cetaceans exist within the literature (Cords and Würsig 2014), with the majority of those records involving singletons of one species associating with groups of another species (*e.g.* Jefferson *et al.* 1991, Bearzi 1996, Baraff and Asmutis-Silvia 1998). There are even fewer suggestions as to why such associations may persist.

Some general trends are evident, however; the formation of heterospecific cetacean groups predominantly occurs in gregarious species (such as delphinids) that normally live together in social groups (Stensland *et al.* 2003). In line with general group theory (Acevedo-Gutiérrez 2008), increased predation avoidance and/or improved foraging success due to larger group size are generally cited as the most likely drivers of mixed-species associations

of cetaceans, with social factors possibly also playing a role (Stensland *et al.* 2003, Cords and Würsig 2014).

4.4.1 Hypotheses regarding the possible functions of associations between false killer whales and bottlenose dolphins in the study area

The almost permanent nature of the observed associations between false killer whales and bottlenose dolphins, poses challenges for testing their functions as it precludes the possibility of relating association frequencies to any possible drivers such as predation risk or foraging behaviour. In the light of these constraints, the most likely functions are discussed to provide the basis for plausible hypotheses.

4.4.1.1 The null-hypothesis

The simplest explanation for the observed interspecific associations between the two species is that they are random encounters between sympatric species due to the independent attraction to environmental features such as feeding grounds (Waser 1982, Würsig and Cords 2014). However, photo-id results suggest that interspecific associations, not only between the two species, but also between individual members of each species, occur repeatedly and at temporal and spatial scales much larger than would be expected from chance or the random independent utilisation of preferred foraging areas. The null hypothesis appears, therefore, unlikely.

4.4.1.2 Anti-predatory benefits

A comparison with other taxa indicates that the correlation between predation pressure and the formation of heterospecific groups appears strong in some primate studies and has been suggested to support the anti-predatory advantages of mixed-species groups (*e.g.* Heymann and Buchanan-Smith 2000, Eckardt and Zuberbühler 2004). While there are few conclusive studies identifying the function/s of mixed-species groups in cetaceans, a few exceptions exist. Norris and Dohl (1980) suggested that associations between spinner (*Stenella longirostris*) and spotted dolphins (*Stenella attenuata*) provided spinner dolphins with safer resting opportunities due to the spotted dolphins' superior alertness. Similarly, Kiszka *et al.* (2011) concluded that spinner dolphins and pantropical spotted dolphin form mixed-species groups for protection against predators when transiting between resting areas. Results presented herein do indicate that anti-predatory factors may play a role in the formation of the mixed-species groups observed in the study area. The predation event observed in the course of this study (Visser *et al.* 2010, Appendix B) shows that killer whale predation is a

bonafide threat to both false killer whales and bottlenose dolphins in New Zealand waters and likely elsewhere.

The increase in group size is perhaps the most obvious effect achieved by these mixed-species associations, potentially facilitating standard anti-predatory advantages such as each individual's decreased likelihood of predation (*dilution effect*) and/or the increased chances of predator detection (*more eyes* approach, e.g. Norris and Schilt 1988). Furthermore, observations show that both species will contract into a tightly intermingled single group in the presence of killer whales, suggesting that a joint approach may be the preferable anti-predatory strategy, although it may simply be a case of cover seeking (Morse 1977). While there are factors that support the potential for increased anti-predatory benefits of mixed-species group formation, they remain speculative in nature.

4.4.1.3 Foraging benefits

Mixed-species groups may also experience foraging advantages, in particular in open habitats (Norris and Schilt 1988). The abundant, yet uneven, occurrence of prey in the open ocean habitat may favour mixed-species associations as resources would be easier to detect (*more eyes*). Concurrently, resources would unlikely be depleted by the larger heterospecific group thus minimizing resource competition; a situation that may apply to many open ocean delphinids. Foraging advantages were suggested by Qu  rouil *et al.* (2008) who concluded that the driver of associations between striped (*Stenella coeruleoalba*), common (*Delphinus delphis*), bottlenose and spotted dolphins in Azorean waters was likely food related, based on the comparatively higher foraging rates observed in mixed-species groups and higher association rates during preferred feeding periods.

Improved foraging, may also apply to the observed associations in this study, as joint feeding appears to be a regular component. Packer and Ruttan (1988) suggest that size and abundance of prey are important determinants of predator cooperation during foraging with small, multiple prey favoring cooperation by its captors. Indeed, joint foraging on large schools of kahawai (*Arripis trutta*) was observed in the study area on several occasions (Zaeschmar *et al.* 2013, Appendix C, refer also to Chapter Two). Given that the prey is found in large schools that are unlikely to be completely depleted by one species alone, the increase in numbers, achieved by the formation of these mixed-species groups may result in greater foraging success due to an increased likelihood of encountering prey. Once encountered, prey may be more effectively captured by the combination of each species' diverging prey capture strategies and skills. Both species were observed feeding on the same

prey species (kahawai), and herding a common prey species in an apparently cooperative manner on several occasions. However, both species were also observed to feed separately. As foraging observations increased with encounter duration (Refer to Chapter Two), it is likely that joint foraging is more prevalent than is currently being recorded.

Another foraging related hypothesis that is also supported by some primate observations is the access to otherwise unavailable food sources for members of heterospecific groups (Struhsaker 1981, Cords and Würsig 2014). While there is some overlap in diet between the two species in the study area (*e.g.* kahawai), false killer whales routinely capture prey considered too large for bottlenose dolphins to ingest. For example, false killer whales were observed to food share large king fish (*Seriola lalandi*) and hapuku (*Polyprion oxygeneios*). While fish of this size appear unlikely to be captured by bottlenose dolphins, it is conceivable that they partake in the feeding, either through food sharing or through social parasitism. Indeed, observations of mixed groups of false killer whales and bottlenose dolphins from Azorean waters document bottlenose dolphins scavenging on food scraps of a large unidentified *Thunnus* being food shared by false killer whales (D. Goetz pers. comm.¹¹). While these observations appear to suggest social parasitism on behalf of the bottlenose dolphin, commensalism may be the more accurate description as the false killer whales are unlikely to be disadvantaged by the loss of the scraps. Furthermore, some display of agonistic behaviour by the false killer whales would be expected if food competition was that direct. However, no aggressive behaviour between the two species has been observed in the study area.

The frequency of foraging observations, including the apparent joint herding of prey, suggests that food may be an important factor behind the association and would support observations from Japanese waters (Tsutsumi *et al.* 1961). However, based on these observations alone, it cannot be ascertained if improved foraging is indeed occurring and if it is a cause or a consequence of the association. More data are required to elucidate the role of prey access in the formation of these mixed-species associations.

4.4.1.4 Social factors

Social factors are another possible function of interspecific associations and have been suggested as the driving factor behind associations between Atlantic white-sided dolphin (*Lagenorhynchus acutus*) and long-finned pilot whale associations (Baraff and Asmutis-Silvia, 1998) and between common and bottlenose dolphin associations (Bearzi 1996),

¹¹ Daniel Goetz, www.danielgoetz.com

based mostly on the apparently permanent nature of the associations. It is worth noting that both the aforementioned examples consisted of singletons of one species associating with groups of heterospecifics. Frantzis and Herzog (2002) suggested that interspecific associations may be particularly frequent if conspecifics are unavailable as social partners based on observations of small numbers of Risso's (*Grampus griseus*) and common dolphins associating with larger groups of striped dolphins in the eastern Mediterranean.

Social factors have also been indicated to contribute towards the formation of groups of bottlenose dolphins and Indo-Pacific humpback dolphins (*Sousa chinensis*) in the western tropical Indian ocean by Stensland *et al.* (1998), based on frequently observed interspecific sexual behaviour and apparent allo-parental care. The prolonged close interspecific interactions, including physical contact together with the high rate of interspecific re-association between individuals observed in the present study, suggest that social factors may also play a role. However, the ephemeral nature of associations between the two species in other regions where they occur sympatrically, such as Hawaii (Baird *et al.* 2008), indicates that sociality may be a function of heterospecific group formation rather than its cause.

4.4.1.4 Costs of association

In line with standard costs of group living (Acevedo-Gutiérrez 2008), increased resource competition has been suggested as the principal potential cost faced by heterospecific cetacean groups (Bearzi 2005). Larger groups are thought to be reproductively disadvantaged due to the higher energetic cost of travel incurred by more rapidly depleted resources (Acevedo-Gutiérrez 2008). However, cost of travel may be offset by delphinids' highly energy-efficient forms of locomotion, which may significantly reduce the energetic burden on its members (Acevedo-Gutiérrez 2008).

While resource competition may be occurring in the observed associations in this study, no obvious indicators were apparent. Consequently, direct food competition may be offset by the abundance of prey once it is encountered. Furthermore, despite overlapping diets, a certain degree of niche segregation is apparent, manifested by the diverging prey species observed in the course of this study. The efficient locomotion of both species (Fish 1998), suggests that the greater energetic cost incurred by the need to cover larger distances to counteract food depletion is likely negligible (Acevedo-Gutiérrez 2008). Additionally, the different body masses between the false killer whale (maximum weight 1,350 - 2,500 kg, Leatherwood *et al.* 1989, Reidenberg and Laitman 2008) and the bottlenose dolphin (up to

650 kg, Jefferson *et al.* 2008) may mean that the two species' ecological requirements are divergent enough to avoid direct food competition (Heymann 1997).

4.4.1.5 Divergence in costs and benefits

Heterospecific groups may not necessarily be mutually beneficial and costs as well as benefits may be experienced differently by participants (Cords and Würsig 2014). For example, associations of small numbers of common dolphins with large groups of dusky dolphins unlikely produces any quantifiable benefits to the dusky dolphins, while the anti-predatory and foraging advantages for the common dolphins may be substantial (Würsig *et al.* 2007). Social parasitism (Norris and Prescott 1961) may also occur in some mixed-species groups and has been suggested as the underlying factor behind the displacement of prey-herding common and pantropical spotted dolphins by bottlenose dolphins (Clua and Grosvalet 2001). Likewise, social parasitism has been inferred from observations of bottlenose dolphins and short-finned pilot whales (Shane 1994); bottlenose dolphins would initiate associations with feeding pilot whales and terminate associations when the whales stopped feeding. The pilot whales were observed to form tighter groups when bottlenose dolphins were present and would change direction and/or move closer to shore; behaviours, which have been interpreted as possible avoidance (Shane 1994).

While the apparent cooperation during some of the joint prey herding observations in false killer whale-bottlenose dolphin associations is indicative of reciprocal mutualism, the nature of these interspecific associations is difficult to determine, based on these observations alone. Although, social parasitism, in the form of one species taking advantage of the other's superior prey locating or capturing abilities, cannot be dismissed (see potential scavenging earlier), the absence of observed agonistic behaviour by either species makes it an unlikely driver. Furthermore, the rate of re-association between individuals of both species as documented in the study area, indicates that individual recognition between heterospecifics is likely. A more sophisticated group structure by the two species may therefore have to be considered. However, commensalism may be occurring and would be difficult to distinguish from mutualism based on these observations alone. Indeed, the aforementioned observations of possible scavenging by bottlenose dolphins in Azorean waters would support a degree of commensalism. Divergence in species specific costs and/or benefits may therefore be evident.

In contrast to the difficulties of identifying possible functions of these mixed-species groups, associations between false killer whales and long-finned pilot whales in the study area

appear less complex. Observations suggest that they may be primarily social in nature based on their short duration, absence of observed foraging behaviour and closeness of interaction. While associations with pilot whales appear to be infrequent and fleeting in the study area, they may occur more regularly in offshore habitats. More observations are required to ascertain the extent and/or nature of associations between false killer and pilot whales.

4.4.2 Conclusion

Results presented herein show that the observed false killer whale-bottlenose dolphin associations in the study area are unlikely to occur at random. In line with standard group theory, plausible indicators for improved predator evasion and foraging success have been identified as potential functions of the observed interspecific associations. Furthermore, while social factors appear to be present, results indicate that they are more likely to be a consequence of association rather than their cause. However, hypotheses presented here may not be mutually exclusive. Consequently, mixed-species groups may experience anti-predatory as well as foraging advantages and social stimulus. Results further suggest that the association is likely complex and may include other, yet unidentified factors. It is important to emphasise that none of the hypotheses generated herein represent critical tests but facilitate greater understanding of the processes required to successfully assess the nature and extent of these intriguing associations. As such, these findings highlight the importance of long-term behavioural data collection and strongly support the need for dedicated research to be conducted on the species in New Zealand waters.

Chapter Five

Conclusions and implications for conservation and management



False killer whales foraging off northeastern New Zealand. Photo © Jochen Zaeschmar

5.1 Introduction

The management and subsequent conservation of cetaceans has witnessed some remarkable changes in the last 50 years. Although historic threats such as commercial whaling (Baker and Clapham 2004) have been largely diminished, these have been replaced by a host of more complex anthropogenic impacts, ranging from the more direct effects of fishery interactions (Read 2008) and vessel collisions (Van Waerebeek *et al.* 2007) to less quantifiable factors such as resource competition, habitat degradation, tourism, noise, pollution and climate change (Reeves *et al.* 2003, Kaschner and Pauli 2005, Weilgart 2007, Higham *et al.* 2008, Simmonds and Elliott 2009, Rolland *et al.* 2012, Simmonds 2012). Assessing the impact that these possible threats may have on species or populations is arguably one of the great challenges faced by marine mammal science today (Lee and Jetz 2011, Davidson *et al.* 2012). Likewise, the inherent difficulties of accurately assessing the status of cetacean populations, in particular those of more cryptic species (MacKenzie *et al.* 2005, Williams and Thomas 2009), make it difficult to ascertain appropriate conservation statuses and management strategies.

The problem is further compounded by the challenge of appropriately defining populations for management purposes as this can greatly affect, not only the respective species, but also the ecosystems they inhabit (Taylor 1997). For example, defining populations purely on genetic grounds invites the risk of jeopardising ecosystem health as it does not address the implications of fragmentation and local extirpation (Taylor and Dizon 1999). Once the need for protection is established, finding appropriate means to effect management is another obstacle as it requires not only the identification of threats but also of critical habitat (Baird *et al.* 2012) of populations whose home ranges are often unknown.

The false killer whale's infrequent yet prominent stranding history in New Zealand (Brabyn 1991), together with its known tendency to interact with fisheries in other regions (*e.g.* Ramos-Cartelle and Mejuto 2007), makes it a species of potential conservation concern. Yet, despite being considered *Data Poor*, the false killer whale is classified as *Not Threatened* in New Zealand (Townsend *et al.* 2008).

This final chapter summarises the findings of this study and compares them to those from other regions. Results are then related to possible conservation issues affecting false killer whales in the study area, with the aim to provide the first baseline data for appropriate management and conservation. Finally, directions for future research are assessed and resulting recommendations presented.

5.2 Summary of findings

- Stranding and sighting records suggest that the species occurs primarily in waters north of the subtropical convergence, which includes the North Island and the Chatham Islands. None of the sightings reported herein and only 7% of documented strandings occurred in South Island waters.
- Seasonal trends in occurrence were apparent, with all at-sea observations off northeastern New Zealand and over 70% of strandings throughout New Zealand, documented between December and May. Occurrence off northeastern New Zealand correlates with the seasonal shoreward flooding of a warm ocean current.
- Site fidelity off northeastern New Zealand is considerably greater than would be expected for a vagrant species, with the vast majority of individuals resighted over several years in several locations along an approximate 650 kilometer (km) stretch of coastline.
- Social network analysis shows that all individuals identified in the study area so far are linked by association in one large albeit apparently clustered social network.
- Abundance estimates suggest that the number of false killer whales that frequented the study area between 2005 and 2012, may be as low as 111 individuals.
- Interspecific associations with presumed oceanic bottlenose dolphins (hereafter referred to as bottlenose dolphins) were observed during the vast majority of encounters off northeastern New Zealand. Findings suggest that these interactions are not random, with individuals of both species resighted together over five years and 650 km apart.
- Foraging observations revealed herding of large schools of coastal fish species in apparent cooperation with bottlenose dolphins. This included predation on a previously undescribed prey species and the use of a yet undocumented bubble technique. However, some of the same individuals were also observed to be feeding on large demersal fish in offshore waters.

5.3 Similarities and differences to false killer whale populations studied in other regions

The general lack of dedicated and ongoing studies on false killer whale population dynamics means that few possibilities for comparisons exist outside of the Hawaiian insular population. The New Zealand and Hawaiian populations share a number of similarities but also show some marked differences: The clustered and closed social network identified in

the present study is consistent with that reported from Hawaiian waters (Baird *et al.* 2008). The majority of individuals of both populations are marked and the rates of mark change are very similar (Baird *et al.* 2005). Likewise, the high level of site fidelity observed in New Zealand waters, reflects that reported for the Hawaiian insular population (Baird *et al.* 2008, Baird *et al.* 2012). Site fidelity in coastal waters has also been reported from Costa Rica and the Gulf of Guinea (Acevedo-Gutiérrez *et al.* 1997, Weir *et al.* 2013). However, unlike the Hawaiian insular population, which can be observed in nearshore waters year round (Baird *et al.* 2012), there appears to be a strong seasonal component in coastal false killer whale occurrence in New Zealand waters. Seasonal occurrence in coastal waters has also been observed in other regions (Findlay *et al.* 1992, Acevedo-Gutiérrez *et al.* 1997, Palmer *et al.* 2009). Observations from Japanese waters suggest a correlation between occurrence in coastal waters and the shoreward flooding of a warm ocean current, similar to that documented in the present study (Kasuya 1985, Kasuya 1971).

Group size estimates in the present study appear to be larger than those usually reported from Hawaii and other warm regions (Acevedo-Gutiérrez *et al.* 1997, Baird *et al.* 2013, Weir *et al.* 2013) and more comparable to those observed in cooler waters off Japan and South Africa (Kasuya 1986, Best 2007, Kirkham *et al.* 2010). Likewise, the almost permanent interspecific associations with bottlenose dolphins observed in New Zealand waters have not been reported from Hawaii but are consistent with reports from other regions, including Japan and South Africa (Mizue and Yoshida 1961, Tsutsumi *et al.* 1961, Best and Reeb 2010). False killer whales in the present study show significantly fewer signs of fishery interactions than the Hawaiian populations (Baird *et al.* 2014a). Meanwhile, the observed frequency and sizes of mass stranding events documented in the present study appears consistent with those reported from other regions (Phillips 1988, Kirkman *et al.* 2010, Ferreira *et al.* 2014).

5.4 Management implications and recommendations

5.4.1 Current conservation measures

The false killer whale is currently classified as *Not Threatened* in New Zealand (Baker *et al.* 2010) despite being considered *Data Poor*. Consequently, the species features only peripherally in New Zealand marine mammal protection considerations. Despite the large number of stranded individuals, actual stranding events remain rare. Furthermore, the infrequency of sighting reports and the apparent lack of interactions with fisheries have given little cause for conservation concern. Due to the stated paucity of data, the false killer

whale is only listed under *other toothed cetaceans* in the Department of Conservation's Marine Mammal Action Plan (MMAP, Suisted and Neale 2004). The MMAP sets out a range of species- and issue-led conservation objectives for marine mammals occurring in the New Zealand region. In the case of the false killer whale, as with all 'other' toothed cetaceans, the aim is to facilitate *effective general public awareness, advocacy and stranding response*, with the key objective being to *better understand the population ecology, key habitat requirements and threats to toothed cetaceans in New Zealand waters*.

The present study provides the first baseline data of false killer whale population ecology in New Zealand waters, directly addressing the MMP's key objective. Furthermore, it provides the necessary information for effective public awareness and advocacy. Results presented herein indicate that the coastal waters off northeastern New Zealand are frequented by a relatively small number of false killer whales. Such a finding has a number of potential implications for the appropriate management of the species in New Zealand, which are discussed below.

5.4.2 The likelihood of a small population and its implications

The population baseline data presented here suggest that the number of individuals frequenting the coastal waters off northeastern New Zealand is small. However, it remains unknown if these individuals form part of a larger primarily oceanic metapopulation or if they are members of a small, local and genetically closed population. Studies from Hawaiian waters show that false killer whales with high site fidelity form part of a small and genetically closed population (Baird *et al.* 2008, Chivers *et al.* 2010). A localised disjunct population in New Zealand waters can therefore not be ruled out. Furthermore, the prospect of a small number of individuals with high site fidelity presents conservation implications regardless of the population's genetic distinctiveness. While a reduction in the numbers of these individuals may not affect the viability of a possible metapopulation, the ecosystem services provided by false killer whales in coastal waters would be reduced through the resulting top-down effects (Taylor 1997, Estes *et al.* 2001). Additionally, a loss of these individuals may lead to fragmentation of the metapopulation (Taylor and Dizon 1999). Consequently, it is recommended that the false killer whales identified in this study also be considered part of a small local population for management purposes regardless of their genetic identity.

5.4.3 Fisheries interactions

False killer whales are known to interact with fisheries to the extent that they are considered a problem species in some regions, with depredation of pelagic long-lines known to cause injury and even mortality (*e.g.* Bargain *et al.* 2000, Gilman *et al.* 2006, Ramos-Cartelle and Mejuto 2007, Hernandez-Milian *et al.* 2008, Baird *et al.* 2014a). While some of the dorsal fin disfigurement observed on false killer whales in this study is consistent with line cuts from fishing gear, only two individuals showed such trauma. Furthermore, both whales exhibited the scarring when first photo-identified and no new injuries have been observed since the beginning of the photo-identification (photo-id) efforts in 2005. As there are no known reports of fisheries interactions in New Zealand waters, it remains unknown if, or to what extent, they occur. However, based on scarring alone, fisheries interactions do not appear to constitute a major cause of injury for the individuals identified in this study. Given that the species is often confused with other globicephalids (*e.g.* Baird 2010, refer also to Chapter Two), more stringent identification of cetaceans interacting with fisheries in New Zealand waters is recommended.

5.4.4 Implications of a mass stranding event

The false killer whale's tendency to strand en masse is also apparent in New Zealand, with some of the country's largest events accredited to the species (Brabyn 1991). Given the apparent scarcity of the species in New Zealand waters and the possibility of a small local population, extensive refloating efforts as well as thorough data collection of stranded individuals are highly recommended.

5.4.5 Interspecific associations with bottlenose dolphins

The frequency of observed associations between false killer whales and bottlenose dolphins in the study area suggests that these interactions are an integral part of the species' respective ecologies. As a result, threats pertaining to one species likely also affect the other. This finding poses the question if or to what extent the two species should be studied and/or managed separately.

5.4.6 Misidentification

Misidentification of false killer whales, at strandings and at sea, continues to be a concern as it may grossly misrepresent the number of individuals that succumb on New Zealand shores or are affected by anthropogenic impacts such as fisheries. Consequently, any reports of false killer whales should be accompanied by good quality media that clearly show the species' distinguishing features (small falcate dorsal fin, torpedo shaped body, lack of saddle patch or

eye stripe). This should be the bare minimum of data collected. In the case of stranded individuals the large conical teeth and lack of white pigmentation around the “lips” should also be documented. Department of Conservation staff in charge of identifying cetaceans at strandings should be aware of the distinguishing features of the different globicephalids, which may strand on New Zealand shores. The production of a detailed ID-guide for strandings is highly recommended. Technological advances in recent years mean that the production and speedy distribution of high quality images is increasingly effortless, which should be helpful for correct and swift species ID of false killer whales and cetaceans in general.

5.4.7 Advocacy and public awareness

The false killer whale remains one of the lesser known cetaceans (Baird *et al.* 2008), with the general public often unaware of the species’ existence, let alone local occurrence. Given the likelihood of a small local population and the issue of species misidentification, increasing public awareness of false killer whales in New Zealand waters is recommended. Additionally, the public appeal of a charismatic apex predator such as the false killer whale (Sergio *et al.* 2008) may make it a suitable flagship species for conservation issues affecting the near or offshore marine environment of New Zealand.

5.5 Study limitations

5.5.1 The benefits and limitations of using platforms of opportunity

This thesis is the result of the collation of governmental stranding records and sighting reports from whale-watch vessels and dedicated research platforms. Consequently, the study highlights both, the possibilities but also the limitations that such an approach presents. Opportunistic observations can yield otherwise unobtainable broad-scale information on the status and occurrence of cetacean species (*e.g.* Kiszka *et al.* 2004). Paired with photo-id they can provide crucial insights into site fidelity, social organisation and home-ranges. One of the greatest benefits of platforms of opportunity is arguably that they offer a low cost means of continuous long-term data collection. This can prove particularly useful in the study of rare or cryptic species such as the false killer whale, which may go completely undetected or significantly under-reported in more sophisticated but more limited surveys. For example, sophisticated short term surveys in the English Channel yielded no harbour porpoise (*Phocoena phocoena*) sightings, yet opportunistic sightings showed that the species was present in low numbers and further indicated more frequent occurrence in western parts and during winter months (Evans and Hammond 2004).

However, there are a number of limitations that have to be considered, especially in the analysis of opportunistic data. The assessment of the New Zealand stranding record revealed some inconsistencies and discrepancies with previously published records and the possibility of species misidentification could not be ruled out completely in many of the records. In regards to the sighting records, the heterogeneity of observation platforms, including differences in boat speed, arc of vision, observer skills, -numbers, -height and/or environmental factors usually associated with the use of platforms of opportunity, made some comparisons difficult. However, these limitations are offset by the temporal and spatial scale of the study, which produced robust and meaningful, albeit broad scale results about the occurrence of false killer whales in New Zealand waters, in particular in northeastern parts. As such, the study demonstrates the benefits that long-term data collection from platforms of opportunity can offer, allowing scholars to gather information that is otherwise effectively unobtainable. The resulting findings enable wildlife managers to make informed decisions on species previously considered too data poor to assess.

5.5.2. Recommendations for opportunistic data collection

The quality of opportunistic data can be greatly improved if a number of guidelines are adhered to:

- Data sets should cover large time scales. As many data from platforms of opportunity lack the prerequisites for fine scale analysis, resulting limitations can be offset by the strength facilitated by long-term data collection.
- The larger the number of platforms/observers and their coverage, the smaller the bias.
- Observer coverage should be extended evenly across as wide an area as possible and during all seasons.
- Effort should be quantifiable. Opportunistic data are often limited by the inability to measure effort. Trip encounter rates offer a basic means to calculate effort but may be subject to sample bias in the absence of dedicated effort track lines.
- Correct species identification needs to be ensured. The use of photographs and/or video footage as proof of presence is therefore essential. Fortunately, ever advancing camera technology means that even untrained observers can produce high quality images of cetaceans in many cases even allowing for the identification of individuals.

5.6 Future Research

5.6.1 Species specific research

This study highlights the need for a species specific study of false killer whales in New Zealand waters. Key research questions should focus on population dynamics, including trends in abundance, home ranges and genetic identity. Are the individuals identified in this study part of a larger oceanic metapopulation or do they belong to a local and possibly genetically closed population? How large is the population that the identified individuals belong to? What are the home-ranges of false killer whales identified in this study? What is the extent and function of the observed associations with bottlenose dolphins? To what extent do false killer whales in New Zealand waters face anthropogenic threats?

In order to elucidate these research questions, it is recommended that dedicated and continued photo-id efforts be extended and behavioural studies instigated to fully assess habitat use. It would be prudent to extend the research to include offshore areas or at least make use of existing observer programmes and/or opportunistic platforms to assess false killer whale distribution within New Zealand waters.

Genetic sampling would answer questions about population identity. Additionally, satellite tagging, for a species such as false killer whales, which are known to frequent offshore waters, would elucidate aspects of home ranges and seasonality (Baird *et al.* 2010). Genetic sampling and tagging are invasive methods with potential detrimental effects on the sampled individual (Schneider *et al.* 1998, Bearzi 2000). However, reports of negative impacts appear to be isolated incidents and stand in marked contrast to the far greater number of sampling efforts without apparent detrimental effects (*i.e.* Martin *et al.* 2006, Noren and Mocklin. 2012). Long-term monitoring of satellite tagged false killer and short-finned pilot whales (*Globicephala macrohynchus*) in Hawaiian waters showed no significant differences in survival (Baird *et al.* 2013a).

Genetic sampling has provided the necessary evidence to identify the Hawaiian insular false killer whale population as genetically closed (Chivers *et al.* 2007, Chivers *et al.* 2010) and was a key determinant of the population's proposed 'endangered' status (Oleson *et al.* 2010, Baird *et al.* 2012). Likewise, satellite tagging has proven that while the Hawaiian insular population is indeed exclusively island associated, it still interacts with the Hawaiian long-line industry (Baird *et al.* 2014a), thought to be the main reason for the population's apparent decline (Reeves *et al.* 2009). These findings subsequently provided the necessary justification to amend fishing practices and to extend the long-line exclusion zone in Hawaiian waters in order to more effectively protect the population (Baird *et al.* 2014a).

Therefore, the potential, but as yet unproven, detrimental effects of the recommended invasive research techniques appear justified in the light of these demonstrable conservation benefits.

5.7 Conclusion

This thesis constitutes the first insight into the population ecology of false killer whales in New Zealand waters. The study is intended as an initial step in the assessment of the species' status in New Zealand, based on stranding records and at-sea observations. Results represented indicate that false killer whales predominantly inhabit North Island waters. While strandings are rare they can be extremely large and challenging events, numbering several hundred individuals. Despite being infrequently sighted at sea, a relatively small number of individuals clearly show site fidelity in waters off northeastern New Zealand, albeit with a strong seasonal aspect. While it remains unknown if these individuals belong to a larger metapopulation or a small local population, studies from Hawaiian waters have shown that socially connected false killer whales with high site-fidelity form part of a small and genetically closed island-associated population. A small localised and disjunct population in New Zealand waters is possible and thus, a precautionary approach in the assignment of an appropriate conservation status may be prudent. Additionally, extensive refloating efforts of live stranded individuals is strongly recommended together with thorough data collection of deceased animals. Due to the risk of confusion with the considerably more common pilot whales, awareness raising measures among wildlife managers and the public are desirable.

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Appendices

Appendix A

Whale and dolphin operator and research platform poll regarding presence/absence of false killer whale sightings in their respective areas of operation

Questions and results of the initial poll of whale- and dolphin-watch operators and dedicated research platforms (*) from around New Zealand. Participants were contacted by email and asked to answer eight questions about their operation and sighting records of false killer whales. Respondents who did have records of false killer whales were contacted again for details such as photographs, exact dates and group size estimates.

Questions:

1. What is your area of operation?
2. What are your years of operation (*e.g.* 2000-2005 or since 1998)?
3. What months of the year do you operate (*e.g.* October – May, year-round *etc.*)?
4. How many vessels do you operate?
5. Do you have any records of false killer whales?
6. If you have seen false killer whales, how many records do you have?
7. If you have seen false killer whales, can you provide the dates of the records?
8. If you have seen false killer whales, do you have any photographs or video footage of the encounters?

Answers:

Operator	1. Area	2. Timeframe	3. Season	4. Number of vessels	5. Records	6. Number of records	7. Dates of records	8. Proof of presence
North Island								
Cascade Charters	Three Kings Is.	2007-2008	Apr-May	1	Yes	5	Y	Y
Dolphin Rendezvous	Doubtless Bay	2004-2009	Sep-May	1	Yes	1	N	N
Carino Dolphin Sail	Bay of Islands	1995	Sep-May	1	Yes	4	Y	Y
Dolphin Discoveries	Bay of Islands	1992	All year	1	Yes	7	Y	Y
Ecocruz	Bay of Islands	2000	Oct-May	1	Yes	4	Y	Y
Great Sights	Bay of Islands	1994	All year	2	Yes	29	Y	Y
Dive Tutukaka	Poor Knights Is.	1999	All year	3	Yes	1	Y	Y
Ocean Blue	Poor Knights Is.	2006	All year	1	Yes	1	Y	Y
Dolphin Explorer	Hauraki Gulf	2002	All year	1	Yes	1	Y	Y
Te Epiwhani*	Hauraki Gulf	2010-2012	All year	1	Yes	1	Y	Y
Aronui Moana*	Bay of Plenty	2010-2012	All year	1	Yes	2	Y	Y
Dolphin Seafaris	Bay of Plenty	2006	Nov-May	1	Yes	Y	Y	Y
Pee Jay White Is. Tours	Bay of Plenty	1995	All year	3	Yes	Y	Y	Y
South Island								
Operator	1. Area	2. Timeframe	3. Season	4. Number of vessels	5. Records	6. Number of records	7. Dates of records	8. Proof of presence
Dolphin Watch Tours	Marlborough Sds	1995	Oct-May	2	No			
Golden Future	Marlborough Sds	2002	All year	1	No			
Dolphin Encounter	Kaikoura	1989	All year	3	No			
Whale Watch Kaikoura	Kaikoura	1987	All year	3	No			
World of Whales	Kaikoura	1990	All year	3	No			
Black Cat Cruises	Banks Pen.	1988	All year	3	No			
Real Journeys	Fiordland	2002	All year	4	Yes	1	No	No
Monarch Wildlife Cr.	Otago Pen.	2002	All year	1	No			
Fiordland Expeditions	Fiordland	2004	All year	2	No			
Aihe Eco charters	Stewart Island	1999-2008	All year	1	No			
Rakiura Charters	Stewart Island	2009	All year	2	No			

Appendix B

VISSER, I. N., J. ZAESCHMAR, J. HALIDAY, A. ABRAHAM, P. BALL, R. BRADLEY, S. DALY, T. HATWELL, T. JOHNSON, W. JOHNSON, L. KAY, T. MAESSEN, V. MCKAY, N. TURNER, B. UMUROA and D. S. PACE. 2010. First Record of Predation on False Killer Whales (*Pseudorca crassidens*) by Killer Whales (*Orcinus orca*). *Aquatic Mammals* **36**: 195-204.

First Record of Predation on False Killer Whales (*Pseudorca crassidens*) by Killer Whales (*Orcinus orca*)

Ingrid N. Visser,¹ Jochen Zaeschmar,² Jo Halliday,³ Annie Abraham,⁴ Phil Ball,⁴ Robert Bradley,⁴ Shamus Daly,⁴ Tommy Hatwell,⁴ Tammy Johnson,⁴ Warren Johnson,⁴ Laura Kay,⁴ Tim Maessen,⁴ Vanessa McKay,⁴ Taira Peters,⁴ Nathan Turner,⁴ Basil Umuroa,⁴ and Daniela S. Pace⁵

¹Orca Research Trust, P.O. Box 402043, Tutukaka, 0153, New Zealand; E-mail: ingrid@orca.org.nz

²P.O. Box 91, Puihia, 0247, New Zealand

³6 Kennedy Street, Opuia, 0200, New Zealand

⁴Care of Orca Research Trust, P.O. Box 402043, Tutukaka, 0153, New Zealand

⁵Oceanomare-Delphis Onlus, Via Gino Marinuzzi 74, 00124 Rome, Italy

Abstract

The first record of killer whale (*Orcinus orca*) predation on false killer whales (*Pseudorca crassidens*) is reported here. On 25 March 2010, a group of 50 to 60 false killer whales, including approximately 15 calves and accompanied by three to five bottlenose dolphins (*Tursiops* sp.), were sighted in the Bay of Islands, New Zealand. Within 30 min, they were approached by a group of approximately eight killer whales. Five false killer whales were attacked, with at least three rammed from below, forcing them out of the water. After 29 min, the killer whales were milling at the surface and feeding on the carcass of a false killer whale calf, possibly the only individual killed. The killer whales had prolific fresh and healed oval wounds, which were attributed to cookie cutter shark (*Isistius* sp.) bites.

Key Words: feeding, predator, prey, mixed-species group, cetacean, killer whale, *Orcinus orca*, false killer whale, *Pseudorca crassidens*, cookie cutter shark, *Isistius* sp.

Introduction

The killer whale (*Orcinus orca*), as a species, is an eclectic feeder, taking a wide range of prey (Grachyov & Mymrin, 1986; Sarti et al., 1994; Similä et al., 1996; Ford et al., 1998; Hatfield et al., 1998; Visser & Bonaccorso, 2003; Jones, 2006; Matkin et al., 2007; Visser et al., 2008). However, in some areas, sympatric populations show different foraging strategies as well as prey preference (foraging ecotypes). Various foraging strategies for the New Zealand population have been described in detail, including removing fish from longlines and cooperative hunting for rays, sharks, and cetaceans, with 14 different

species of prey recorded in their diet (Visser, 1999a, 1999b, 2000a, 2000b, 2005; Visser et al., 2000).

Although killer whales predate on cetaceans in various locations around the world (Jefferson et al., 1991; Ford & Ellis, 1999; Visser, 1999a; Saulitis et al., 2000; Ford et al., 2005; Steiger et al., 2008), and accounts of attacks or harassment have been reported for at least 23 different species (Jefferson et al., 1991; Visser, 1999a, 2000b; Pitman et al., 2003; de Oliveira Santos & Netto, 2005), such observations are relatively infrequent and generally only a small number of detailed descriptions of hunting tactics are available. False killer whales (*Pseudorca crassidens*) have not previously been recorded as a prey species for killer whales (Baird, 2008). Hereafter, scientific names will be used when referring to both individual animals and the species in question in order to reduce confusion between the similar common names “killer whale” and “false killer whale.”

Materials and Methods

The observations reported here occurred to the east of the Black Rocks in the Bay of Islands, Northland, New Zealand, at approximately 35°13' S, 174° 09' E. The general area of the encounter is bounded on three sides by various islands and is open to the NE to the Pacific Ocean. Water depth in the area where the attacks occurred is between 30 to 40 m.

All but the first and last authors witnessed various stages throughout the events reported here, and they were either crew or guests aboard one of the seven vessels present. These vessels are tourist dolphin/whale watching boats, details of which are *Tutumui* (length overall [LOA] 11 m, observer deck height [OH] 1.4 m), *Dolphin Seeker* (LOA 22.5 m, OH 3.7 m), *Discovery III* (LOA 11.9 m,

OH 2.5 m), *Discovery IV* (LOA 18.9 m, OH 3.9 m), *Tangaroa* (LOA 19.7 m, OH 3.2 m), and *Carino* (LOA 15.2 m, OH 1.4 m). The second author (JZ) was present in a private vessel *Manawanui II* (LOA 4.1 m, OH 1 m).

In the Bay of Islands, it is standard operating procedure, as part of a Voluntary Code of Conduct, for dolphin/whale watching vessels to shut down their engines when *O. orca* are known to be in the proximity of other cetacean species (Halliday, unpub. data). Therefore, after first approaching the area, the position of each vessel, in relation to the moving animals, varied from a few meters to over 1 km. Regardless of proximity, the observations of the author(s) closest to the event were used, while the additional (distant) observations were used to corroborate the primary (closest) observations. The observations took place when sea conditions were Beaufort 2 or 3.

Times reported here are based on the time codes embedded in the meta-data of digital photos and video collected from observers on the vessels. As these time codes were not synchronised, events were reconstructed based on the authors' observations in conjunction with the noncontiguous video recordings. Timings are approximate and rounded to the nearest minute.

Group composition of the *O. orca* was categorized by age/sex based on the features described for the species (Heyning & Dahlheim, 1988). We used four categories: (1) adult male (distinguishable by the large dorsal fin), (2) adult female (determined by the constant association of a juvenile), (3) juvenile (less than half the size of an adult female and observed to feed on solid food), and (4) uncategorised (adult females and subadult males). Because it is not always possible to distinguish between these two age/sex classes, unless their urogenital pigmentation is viewed (Bigg et al., 1987), both are included in category 4. There were no animals smaller than juveniles (i.e., calves).

Despite a number of mass strandings of *P. crassidens* on the New Zealand coast (Baker, 1981; Robson, 1984), data such as group composition and average length are unavailable for this population. However, as the presence of both sexes and a range of age classes appear to be common in *P. crassidens* groups (Stacey et al., 1994; Chivers et al., 2007; Ferreira, 2008), for the purpose of this report, and based on the observed range in body size, we assume that this was also the case in the group encountered. As young were seen swimming in both the echelon and infant positions (Mann & Smuts, 1999; Gibson, 2006) and, given that we could not easily distinguish between calves and juveniles, we refer to any *P. crassidens* less than half the size of presumed adults as calves, where

an average adult size is 5 to 6 m (Stacey et al., 1994; Ferreira, 2008).

Although bottlenose dolphins (*Tursiops truncatus*) are regularly observed in the coastal waters of the Bay of Islands (Hartel, 2010), those observed with the *P. crassidens*, and reported here, are readily distinguishable from the coastal bottlenose dolphins based on gross morphology. They are comparatively much larger, with a more robust body, and many bear fresh and/or healed oval scars (Zaeschmar, unpub. data), which are presumed to originate from cookie cutter sharks (*Isistius* sp.) (Jones, 1971). These scars have not been observed on the coastal bottlenose dolphins (Halliday, unpub. data). Additionally, the bottlenose dolphins accompanying the *P. crassidens* are considered pelagic in that they are not typically observed close to the coastline (Zaeschmar & Halliday, unpub. data). No morphological, behavioural or genetic research has been done on this pelagic population, so herein they are considered as *Tursiops* sp.

Photo-identification catalogues for the *O. orca*, *P. crassidens*, and *Tursiops* sp. are held by the first two authors. Standard photo-identification techniques for each species (e.g., see Bigg et al., 1987; Würsig & Jefferson, 1990; Baird & Gorgone, 2005) were used to match any photographs taken during the events reported here, to these catalogues.

Results

On 25 March 2010 at 0940 h, a group of between 50 to 60 *P. crassidens* were sighted at Te Nunuhe/Whale Rock (35°12' S, 174°12' E) in the Bay of Islands. There were approximately 15 calves in the group. When observers were within 10 m of the *P. crassidens*, vocalisations could be clearly heard above the water, without the aid of hydrophones. Accompanying the *P. crassidens* were three to five adult *Tursiops* sp. This mixed-species group was travelling in a general WNW direction at approximately 10 to 15 km/h and was followed by two dolphin/whale watching vessels (*Tutunui* and *Discovery III*). A further two dolphin/whale watching vessels (*Discovery IV* and *Dolphin Seeker*) were approximately 2 km away from the *P. crassidens*, with the remaining vessels at farther distances, but all heading towards the group.

Three of the *O. orca* involved in the events reported here were subsequently matched to the *New Zealand Orca Identification Catalogue* (NZOIC) (Catalogue Numbers NZOP-002, NZOP-003, NZOP-004 [all uncategorised], and NZOP-005 [adult female]; Orca Research Trust, unpub. data).

Within 30 min of the *P. crassidens* first being sighted, a group of approximately eight *O. orca*, including one adult male, two juveniles, one adult

female (newly entered into the catalogue as NZOP-006; Visser, unpub. data), and four uncategorized animals, travelling slightly faster than the vessels, were seen approaching from behind *Tutunui* and *Discovery III*. The *O. orca* followed the mixed-species group for approximately 200 m and, while still some distance behind them (at least 500 m), dove below the surface. Concurrently, the mixed-species group dove simultaneously and, upon surfacing, began fast milling in a tight group, thereby creating an agitated water surface. From subsequent observations as well as the number of individuals that next surfaced in this mixed-species group, it appeared that at this point a number of individuals broke away from the main group as approximately 20 to 30 *P. crassidens* with one or two *Tursiops* sp. remained. The other members of the mixed-species group were not seen again.

At 1008 h, and following this period of milling, the mixed-species group began accelerating in a SW direction, porpoising clear of the water at high speed (estimated at more than 20 km/h based on video). At least one *O. orca* approached the mixed-species group underwater and rammed an adult *P. crassidens* from below, forcing it out of the water. This *P. crassidens* was travelling at the trailing edge of the mixed-species group. The *O. orca* also left the water, allowing us to establish that it was not an adult male or juvenile. It was not apparent if the *P. crassidens* escaped. Within that same minute, a *P. crassidens* calf was rammed from below and forced upwards—in this instance, to approximately 2 m clear of the water. During this time, the other *O. orca* had increased speed to approximately match the speed of the *P. crassidens*, and some of the *O. orca* group were travelling close to and parallel with the mixed-species group. The adult male *O. orca* was not observed during this period.

The third attack (at 1008 h) involved two *P. crassidens* simultaneously leaving the water. One, a presumed adult (based on its size, comparative to an *O. orca*), appeared to leave the water voluntarily, with its head pointed skywards (Figure 1a). An adult female *O. orca* (NZOP-006) followed closely and launched itself at the adult *P. crassidens* at such speed that the *P. crassidens* was knocked upwards and its body contorted at the impact site (Figure 1b). This ramming rotated the *P. crassidens* in a tail-over-head direction (Figure 1c). The force of this ramming also propelled the *P. crassidens* further into the air so that its head was approximately 8.5 m (and the tail approximately 12.5 m) clear of the water at the apex of the curve. These height estimates are based on the average body lengths of the two species.

The *O. orca* (NZOP-006) completely left the water in a high arch (clearing the water by approximately 4 m) and re-entered the water while the adult

P. crassidens was still airborne. The *P. crassidens* continued to rotate (Figure 1d) and re-entered the water tail first. This *P. crassidens* was observed to then swim off towards the mixed-species group, which was at that stage approximately 50 m away.

The *P. crassidens* simultaneously involved in this attack was a calf. Figure 1a shows the calf already in the air and upside-down. No *O. orca* was seen attacking the calf above the water, nor below when it re-entered the water. However, given that the calf followed a similar trajectory path as that observed for the adult *P. crassidens* in Figure 1, it may have been rammed while still underwater.

The fourth attack (but involving the fifth *P. crassidens*) was made at 1009 h on a single adult *P. crassidens*, while the mixed-species group continued to move at high speed in a southerly direction towards Tapeka Point (35° 14' S, 174° 07' E). The *P. crassidens* made a similar rotating pivot as the adult in Figure 1; however, we could not ascertain if this *P. crassidens* was directly hit by an *O. orca* despite one (which was not an adult male or juvenile) porpoising clear of the water.

This final attack was made on an adult *P. crassidens*, which was at the trailing edge of the mixed-species group, as the group fled. Following this, the mixed-species group accelerated for 15 s (timed from video), and then all dove simultaneously. Two *P. crassidens* then surfaced, turned, and were swimming towards the *O. orca* that were milling at the surface, while one uncategorized *O. orca* held the *P. crassidens* calf (from attack 3) in its mouth. The two *P. crassidens* approached within approximately 10 m of the *O. orca*, then at least one, possibly both, *P. crassidens* remained in close proximity to the *O. orca* until at least 1013 h, after which they were not seen and may have returned to the mixed-species group. The mixed-species group (containing at least four calves) then turned and headed SE at approximately 10 to 15 km/h towards Robertson Island (35° 14' S, 174° 09' E).

The *O. orca* continued to hold the calf until at least 1014 h, during which time it was still alive. The *P. crassidens* calf was then passed among various members of the *O. orca* group, including to both of the juveniles and the adult male, each of which carried it in its mouth while swimming at the surface. By 1043 h, the *P. crassidens* calf was presumed to be dead while it was carried in the mouth of one of the *O. orca* (uncategorized age/sex) as a small amount of blood was seen and, shortly thereafter, unidentified internal organs were seen through the water surface. Subsequently, both juveniles and the female *O. orca* (NZOP-006) were seen to food-share by feeding from the carcass at the same time, as well as by passing the carcass between them. The *O. orca* not involved in consuming the *P. crassidens* calf were milling and diving in the area. After



Figure 1a. Sequence of events of ramming two *Pseudorca crassidens*; a calf rotates through the air, while an adult starts to breach from the water. Photo © Tommy Hatwell/Explore Images



Figure 1b. An *Orcinus orca* (NZOP-006) rams the adult *P. crassidens*; note the body contortion from the impact. The calf continues to rotate through the air. Photo © Tommy Hatwell/Explore Images

approximately 10 min of food sharing and diving, the *O. orca* headed north at approximately 9 km/h.

During the period in which the *O. orca* were food sharing, the mixed-species group, upon reaching the coastline of the island, began traveling close to the shore at less than 10 km/h. At approximately 1050 h, once they reached Whale Rock, the group increased speed to approximately 20 km/h and headed due north. They were followed for at least 45 min by JZ, during which time they

were within one body length of each other, until they slowly began dispersing, covering an area of approximately 500 × 500 m.

Three *P. crassidens* from the mixed-species group have been matched to a photographic catalogue established in 2005 (Catalogue Numbers NZPC-027, NZPC-034, and NZPC-054) (Zaeschmar, unpub. data). No matches were made for the *Tursiops* sp. (photographic catalogue established in 2005; Zaeschmar, unpub. data).



Figure 1c. The *O. orca* is almost at the apex of its curve, while the adult *P. crassidens* continues upwards. The calf is about to re-enter the water. Photo © Tommy Hatwell/Explore Images



Figure 1d. The *O. orca* and the adult *P. crassidens* are both past the apex of their curve, with the *O. orca* about to re-enter the water and the *P. crassidens* pivoting to re-enter tail first. Photo © Tommy Hatwell/Explore Images

Of note is the number of fresh cookie cutter shark bites on the *O. orca* reported here. The *O. orca* in Figure 1 (NZOP-006) had at least 10 fresh bites on its left side, while the others also had numerous bites on their bodies. Another *O. orca* (NZOP-007) (Figure 2) had at least two fresh wounds on the dorso-thorax area as well as a further two on the caudal peduncle (not shown here). Examination of high-resolution photographs of this animal shows at least five healed scars on the right saddle patch

and 10 healed wounds of similar size and shape on the dorso-thorax area (a total of 19 fresh and healed wounds on the right side).

Discussion

Jefferson et al. (1991) recorded a non-aggressive interaction in the 1970s between a single *P. crassidens* and a group of *O. orca* in the waters of Alaska. There is also one non-aggressive record

of *P. crassidens* and *O. orca* in the waters of the Bay of Islands in April 2005, during which the *P. crassidens* (accompanied by *Tursiops* sp.), upon detecting the *O. orca*, made an extreme course change and travelled away at high speed (Zaeschmar, unpub. data).

The tactics employed by both the predator and the prey in events such as those described here are complex, and it can be difficult to definitively disassociate predation pressures from other ecological aspects; therefore, a single event such as this is not suitable for drawing conclusions. However, it has been proposed that one of the driving forces behind group living is predation (Morse, 1977; Norris & Dohl, 1980; Inman & Krebs, 1987). It is recognised that cetaceans under attack use anti-predator tactics which are species and circumstance dependent, including “flight” or “fight” strategies (Connor, 1996; Heithaus, 2001; Ford et al., 2005; Srinivasan, 2009). Typical responses when adopting the “flight” strategy are coalescence, high-speed directional flight (as a group and dispersing from the group), and movement into shallow waters for refuge (Jefferson et al., 1991; Connor, 1996; Ford & Ellis, 1999; Visser, 1999a).

Coalescence is a common response to stress or danger in many species of cetaceans (Jefferson et al., 1991) and was observed during the events reported here. However, Jefferson et al. (1991) also suggest that this may sometimes result more

from herding by *O. orca* than from defensive manoeuvres by the prey. As the *O. orca* were seen to swim parallel to the mixed-species group after they had formed a tight group, this formation may have resulted from both defence and herding.

O. orca is the fastest of the odontocetes measured and can sprint at 12.5 m/s^{-1} (Williams, 2008) and possibly up to speeds of 15.4 m/s^{-1} (Lang, 1966). Top swimming speeds of *P. crassidens* are unknown; however, animals have been recorded at 7.46 m/s^{-1} (Fish, 1998) and 8.0 m/s^{-1} (Rohr et al., 2002).

The presence of calves/juveniles may have influenced the behaviour of not only the predator but also the prey. The *P. crassidens* may not have been able to flee at full speed. Given that *O. orca* are the fastest of the odontocetes, it is unlikely that young *P. crassidens* could maintain the speed required to evade an attack. The *O. orca* may have been employing a similar strategy they are known to use on tuna—an endurance-exhaustion technique (Guinet et al., 2007). Additionally, given that *P. crassidens* are typically a pelagic species, their response by travelling close to the shore after the attack may be a strategy to avoid further predation—a tactic that is known to be used by grey whales (*Eschrichtius robustus*) travelling with calves (Baldrige, 1972).

Although only two of the ramblings involved calves, it is possible that calves were the primary targets of the *O. orca*. Given the number of calves



Figure 2. Two oval shaped and crater-like fresh wounds on the dorso-thorax and two healed dark oval scars on the saddle patch of one of the *O. orca* (NZOP-007) involved in the *P. crassidens* attack; these wounds and scars are likely from cookie-cutter shark (*Isistius* sp.) bites. Photo © Robert Bradley/Magic Memories

observed in the group of *P. crassidens*, we speculate that the attacked cluster of individuals (which separated from the original group) may have been a nursery group. After the attack, in which at least one calf was killed, the mixed-species group still contained at least four calves. Similar nursery group structure has been observed in other delphinid species in New Zealand waters (Würsig et al., 2007; Stockin et al., 2008).

P. crassidens are considered to be extremely social and affiliative, typically travelling in groups of 20 to 100 individuals (Baird, 2008) and forming strong bonds (as documented by mass strandings) (Caldwell et al., 1970; Phillips, 1988; Stacey et al., 1994). As observed in other odontocete species, maternal care in *P. crassidens* is prolonged and intense, with a high investment by the mother (Hanel, 1986; Mann & Smuts, 1998; Ferreira, 2008). Two *P. crassidens* turned and approached the *O. orca* while the *O. orca* held a live *P. crassidens* calf in their mouths. This may be indicative of a mother or another individual attempting to help the calf. Whether this behaviour was altruistic, epimeletic, or nurturant is open to interpretation, but assisting distressed conspecifics has been recorded for a wide range of cetacean species since the early days of cetology (Caldwell & Caldwell, 1966; Lodi, 1992). Alternatively, this may have been a case of predator investigation (Connor, 2000) by a related or unrelated *P. crassidens*.

The apparent voluntary leaping observed during the third attack (Figure 1) may have been an attempt by the *P. crassidens* to avoid an underwater attack, or the *P. crassidens* may already have been rammed underwater. The speed with which the *O. orca* propelled itself out of the water, following this *P. crassidens*, allowed it to make a substantial physical impact into the side of the *P. crassidens*. Figure 1b shows the distortion of the body of the *P. crassidens*, which, despite anecdotal reports of this impact “snapping her back”,¹ could not be verified and, given that one of the authors (TM) observed this *P. crassidens* swim off to join the mixed-species group, it seems unlikely that its back was broken. It cannot be ruled out that other physical damage (which may or may not ultimately have been fatal) occurred.

It was not possible to ascertain if the attacks were made on the same or different *P. crassidens*, except for the instances where there were gross body size differences (i.e., between adult and calf). Additionally, it is possible that further attacks on the same (or different) individuals were made underwater and thereby were not observed. One tactic of the predator may have been to repeatedly

attack the same individual in an attempt to weaken it. Predation on younger (smaller) individuals may involve a lower energetic cost and/or less risk to the predator than predation on older (larger) animals.

It is likely that at least two *O. orca* attacked individual *P. crassidens* in the group simultaneously (i.e., an adult and a calf *P. crassidens*). Given the body position of the *P. crassidens* calf (Figure 1a) and the similar rotational pivot it takes to the adult in the sequence, it is possible that this calf was knocked from the water by a second *O. orca*, which was not seen to surface beneath the calf. Video establishes a time-line which also indicates this.

Several studies of *O. orca* predation on marine mammals note that females and subadults were active participants throughout the attacks, but mature males had little or no involvement (Arnbom et al., 1987; Silber et al., 1990; Jefferson et al., 1991). Here the attacks were conducted by individual(s) of adult female/subadult male size, with the adult male only participating during the prey-sharing phase. Although Berta et al. (2006) define *O. orca* as having the optimum shape in terms of “fineness ratio” (p. 224) and streamlining, Ford et al. (2005) have suggested that the larger body size and appendages of adult males are likely to have a negative impact on their speed and manoeuvrability compared to the smaller females (and we additionally recognize that such constraints should not affect subadult males).

If fleeing is not a viable option to avoid predation from *O. orca*, adult *P. crassidens* may be physically capable of defending themselves, particularly if they remain in a group. Cetaceans that tend to resist attacks may be more likely to survive attacks than those with more submissive physical reactions (Steiger et al., 2008), certainly when compared to species which are less able to fight back (e.g., *Balaenoptera acutorostrata* [Minke whales]; Ford et al., 2005; Ford & Reeves, 2008).

P. crassidens may resist attacks by predators as they themselves have been recorded harassing cetaceans, including species much larger than themselves—for example, sperm whales (*Physeter macrocephalus*) (Palacios & Mate, 1996) and humpback whales (*Megaptera novaeangliae*) (Weller, 2002). Immediately following the events reported here, one of the *O. orca* (NZOP-003) was photographed with fresh tooth-rake marks on its caudal peduncle. It is not possible to ascertain if they were conspecific rake marks or if they had been inflicted by a *P. crassidens* during the attacks. Given the robust dentition of *P. crassidens* (Stacey et al., 1994), there could be substantial risk for a predator attacking this species.

The three *O. orca* matched to the NZOIC had previously only been seen once (August 2008). They were part of a group of seven *O. orca* which

¹Telegraph News, UK, *Killer whale attacks dolphin in front of tourists* (telegraphic.co.uk, 28 May 2010)

attacked and killed three bottlenose dolphins (coastal ecotype) in almost the exact same location as the *P. crassidens* attacks reported here in the Bay of Islands (Zaeschmar, Halliday, & Visser, unpub. data). The number of fresh and healed cookie cutter shark bite marks on these *O. orca* is of note (e.g., see Figure 2). The *O. orca* which frequent New Zealand waters do not typically have prolific cookie cutter shark bite wounds. Visser (1999c) reported a group of *O. orca* near the Bay of Islands, which had fresh and healed cookie cutter shark bite wounds, but these were apparently from a different population of *O. orca* to the New Zealand population. Cookie-cutter sharks tend to show a distribution with a preference for deep tropical to temperate waters (Bass et al., 1976; Compagno, 1984; Jahn & Haedrich, 1988; Cox & Francis, 1997). The extensive number of bites on the individuals reported here may be indicative that this particular group of *O. orca* has a home range which is centred further offshore and perhaps in warmer waters.

Three *P. crassidens* from the mixed-species group have been matched to a photographic catalogue (Zaeschmar, unpub. data). All three were first photographed in 2007 in the Bay of Islands and were re-photographed on the day of the events reported here—both before and after the attacks (Zaeschmar, unpub. data). No matches were made for the *Tursiops* sp. from the mixed-species group (Zaeschmar, unpub. data).

It is not clear if the presence of the vessels had any role to play in the events described here; however, it is known that boat noise can mask *O. orca* calls (Nowacek et al., 2007). Off Kaikoura (42° 25' S, 173° 41' E), New Zealand, *O. orca* occasionally approach dusky dolphin (*Lagenorhynchus obscurus*) groups in the stern wake of vessels, perhaps using the engine noise (and/or possibly cavitation from the propeller) to mask their presence (Visser, unpub. data). The Voluntary Code of Conduct used in the Bay of Islands by the dolphin watching vessels includes shutting down engines whenever *O. orca* are encountered in the vicinity of other cetaceans in the hope of mitigating any impact the boats may have.

Predator-prey interactions involving cetaceans tend to be difficult to observe and are thus poorly understood. Hence, observations on the effectiveness of the predatory tactics of *O. orca* and of the anti-predator responses of their cetacean prey, such as those reported here, could be helpful as a means to interpret the subtlety of such complex interactions. From a biodiversity or conservation perspective, a better understanding of the dynamics of these predator-prey interactions would help increase our knowledge of both the predator and the prey, thereby allowing us to make better management decisions.

Acknowledgments

We would like to thank all the additional folks who contributed data and information about this event, including Gemma Langsdale, Jo Berghan, Sarah Dwyer, and Samantha Young and two anonymous reviewers made constructive comments on the manuscript. Dagmar Fertl provided access to many references. Rob Hunt from Explore Images and Sue Hews from Magic Memories provided access to images and video. Pam Kerville provided additional video footage. The Orca Research Trust provided financial support for INV to publish this manuscript.

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Appendix C

ZAESCHMAR, J. R., S. L. DWYER and K. A. STOCKIN. 2013. Rare observations of false killer whales (*Pseudorca crassidens*) cooperatively feeding with bottlenose dolphins (*Tursiops truncatus*) in the Hauraki Gulf, New Zealand. *Marine Mammal Science* **29**: 556-562.



Notes

MARINE MAMMAL SCIENCE, **(*): ***_*** (** 2012)

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DOI: 10.1111/j.1748-7692.2012.00582.x

Rare observations of false killer whales (*Pseudorca crassidens*) cooperatively feeding with common bottlenose dolphins (*Tursiops truncatus*) in the Hauraki Gulf, New Zealand

JOCHEN R. ZAESCHMAR,¹ SARAH L. DWYER and KAREN A. STOCKIN, Coastal-Marine Research Group, Institute of Natural Sciences, Massey University, Private Bag 102 904, North Shore MSC, New Zealand.

Multi-species associations of odontocetes remain poorly understood. While their occurrence is occasionally reported, the driving factors behind these interactions are often difficult to ascertain. False killer whales (*Pseudorca crassidens*) are known to associate with a number of other odontocete species, in particular the common bottlenose dolphin (*Tursiops truncatus*) (*e.g.*, Leatherwood *et al.* 1989). However, little is known about the function or extent of these associations.

False killer whales are poorly documented large delphinids. Gregarious by nature, they typically travel in groups of 20–100 individuals and are known to form strong social bonds (Baird *et al.* 2008). Their diet consists primarily of a range of cephalopod and pelagic fish species (Odell and McClune 1999) and may vary by region (Reeves *et al.* 2002). While little is known about their foraging behavior, they have been observed to forage cooperatively in subgroups (Acevedo-Gutiérrez *et al.* 1997). False killer whales occur worldwide in tropical and warm temperate waters (Odell and McClune 1999), with occasional sightings in cold temperate waters (*e.g.*, Stacey and Baird 1991). Their distribution is predominantly oceanic, but they are known to approach close to shore at oceanic islands (*e.g.*, Baird *et al.* 2008) and to occasionally venture into shallow inshore waters (Palmer *et al.* 2009).

Little is known about false killer whale occurrence or distribution in New Zealand waters, with only a few at-sea documented records (*e.g.*, Visser *et al.* 2010). False killer whales occasionally strand around the New Zealand coastline (Baker 1981), although such events typically involve mass strandings (Te Papa Tongarewa, New Zealand Cetacean Stranding Database). To date, there are few data documenting the occurrence of this species in New Zealand waters.

¹Corresponding author (e-mail:jzaeschmar@hotmail.com).

Here we report on two observations of joint foraging by false killer whales and common bottlenose dolphins (hereafter referred to as bottlenose dolphins) in the Hauraki Gulf, New Zealand, in January 2011. The Hauraki Gulf (approximate position 36°10'–37°10'S, 174°40'–175°30'E) is a shallow (depth <60 m), semi-enclosed body of temperate water situated on the east coast of New Zealand's North Island.

On 20 January 2011, crew aboard the MV *Dolphin Explorer*, a local whale and dolphin watching vessel, encountered a group of approximately 150 false killer whales at 36°34.73'S, 174°58.26'E in 43 m of water. The whales were observed in association with a group of approximately 150 bottlenose dolphins, assumed to be the offshore ecotype (Baker *et al.* 2010) based on the prevalence of oval body scars presumed to be the result of cookie cutter shark (*Isistius* sp.) bites (Dwyer and Visser 2011), as well as morphological differences manifested by a more robust body form. Both species were interspersed into mixed-species subgroups over an area of approximately 4 km². Each mixed species group was foraging as indicated by leaps, asynchronous dives and the association of sea birds (Acevedo-Gutiérrez *et al.* 1997).

Nondirectional feeding, as characterized by back and forth movements of individuals heading in different directions and diving asynchronously (Acevedo-Gutiérrez *et al.* 1997) was also recorded, with some false killer whales and bottlenose dolphins observed with unidentified fish in their jaws. Large numbers of flesh-footed shearwaters (*Puffinis carneipes*) were present during the encounter, with many scavenging on floating fish remains. One of these mixed-species subgroups was approached by the observing vessel for closer inspection. This mixed-species subgroup was observed driving a shoal of kahawai (*Arripis trutta*) to the surface (Fig. 1), encircling them using a carouseling technique as described for bottlenose dolphins by Bel'kovich *et al.* (1991). Additionally, bubble bursts were emitted to presumably herd the fish into a condensed "bait ball" (Fig. 2). During this period, the fish were driven closer to the observing vessel with the hull eventually acting as a barrier that restricted the shoal's horizontal movement (Fig. 3) effectively using a wall-formation technique (Bel'kovich *et al.* 1991). Despite efforts to slowly maneuver the vessel away from the bait ball, the mixed-species group continued to restrict the shoal against the hull, while taking turns to feed on individual fish. Several false killer whales were observed with kahawai in their mouths (Fig. 1 insert) and some bottlenose dolphins carried unidentified fish remains presumed to be kahawai also. While both species could be seen to encircle the shoal (Fig. 1) only false killer whales were actually observed ingesting kahawai. Additionally, despite both species being engaged in the herding of the fish, a certain degree of segregation between the two species was observed. Conspecifics were often in clusters of five to eight individuals and less than a body length apart, although generally two to three body lengths away from members of the other species. However, on occasion heterospecific group members were less than one body length apart. The encounter was terminated after 1 h of observation due to time restrictions, with foraging by both species recorded throughout the entire duration. The mixed-species group remained in the area where they were initially encountered and continued to forage.



Figure 1. A false killer whale (*Pseudorca crassidens*) and a bottlenose dolphin (*Tursiops truncatus*) encircling a shoal of kahawai (*Arripis trutta*) using a carouseling technique and (insert) a false killer whale holding a kahawai in its jaws in the Hauraki Gulf, New Zealand on 20 January 2011. Photo by Sarah Gardner.

The second encounter occurred on 25 January 2011 when a mixed-species group of approximately 30 false killer whales and 60 bottlenose dolphins was sighted at $36^{\circ}31.45'S$, $175^{\circ}06.00'E$ in 45 m of water by researchers aboard RV *Te Epiwhania*. The group, which included juveniles of both species, was travelling in a southerly direction as indicated by their consistent movement and short constant dive intervals, and was separated into approximately six single-species subgroups dispersed over approximately 3 km^2 . Initially, the entire group was traveling at *ca.* 10 km/h for a distance of 4 km, after which all individuals altered course to a westerly direction and traveled a further 3 km before slowing and changing behavioral state to milling, as indicated by no net movement, short dive intervals and surfacing facing different directions. Approximately 5 min later, both species were observed foraging together in association with flesh-footed shearwaters that were again observed feeding on the fish remains. Subsequently, a single-species subgroup of false killer whales were closely observed foraging, with other single species groups remaining dispersed over approximately 2 km^2 . Afterwards, a single species subgroup of bottlenose dolphins was recorded using the carouseling and bubble burst technique described during the first encounter, to herd a shoal of kahawai, with individuals taking turns to swim through the bait ball to feed. Throughout the 45 min period of feeding observations all individuals slowly tracked north, with occasional short periods of milling recorded between foraging bouts. For bottlenose dolphins, the herding of fish was visible from the surface, with the prey species identified as kahawai. However, prey species could



Figure 2. Bubble bursts used by a mixed-species group of false killer whales (*Pseudorca crassidens*) and bottlenose dolphins (*Tursiops truncatus*) to herd a shoal of kahawai (*Arripis trutta*) into a tight “bait ball”. Hauraki Gulf, New Zealand, 20 January 2011. Photo by Cathy Goeldner/Explore Images.

not be determined for false killer whales despite observations of fish in their mouths. A certain degree of segregation between species, as outlined previously, was again observed. The encounter was terminated after 2 h when the research vessel departed to rejoin a predetermined survey track. Upon departure, the mixed-species group continued to forage whilst slowly moving in a northerly direction.

Photo-identification of false killer whales was conducted during both encounters, with individuals entered into the New Zealand False Killer Whale Identification Catalogue (JRZ, unpublished data). Twenty five and 13 whales were identified during the first and second encounters, respectively, with at least eight individuals in common between the two encounters.

False killer whales are known to associate with other cetaceans and have been observed in close, nonaggressive association with a number of species including rough-toothed dolphin (*Steno bredanensis*) (Leatherwood *et al.* 1989), Risso’s dolphin (*Grampus griseus*) (Best and Reeb 2010), pantropical spotted dolphin (*Stenella attenuata*) (Miyazaki and Wada 1978), short-finned pilot whale (*Globicephala macrorhynchus*) (Flores *et al.* 2003) and in particular, bottlenose dolphin (*e.g.*, Flores *et al.* 2003, Best and Reeb 2010). Despite these records, only very few observations of false killer whales foraging in mixed-species associations with other cetaceans have been documented (Tsutsumi *et al.* 1961). While the extent and nature of these associations remain poorly documented or understood, joint foraging—as observed during these encounters—may play a role in the formation of these interspecific groups. Standard



Figure 3. A shoal of kahawai (*Arripis trutta*) being herded against the hull of the observation vessel by false killer whales (*Pseudorca crassidens*) and bottlenose dolphins (*Tursiops truncatus*) in the Hauraki Gulf, New Zealand on 20 January 2011. Photo by Sarah Gardner.

benefits of group formation in cetaceans may also apply in this case (Norris and Schilt 1988). Given that the prey are found in large schools that are likely not to be completely depleted by one species alone, the increase in numbers, achieved by the formation of these mixed species groups is likely to result in greater foraging success due to an increased likelihood of encountering prey. As both species were observed feeding on the same prey species, albeit not during the same encounter, and herding a common prey species in an apparently cooperative manner, the association appears to be mutualistic. Packer and Rutan (1988) suggest that size and abundance of prey are important determinants of predator cooperation during foraging with small, multiple prey favoring cooperation by its captors further supporting mutualism. However, based on these observations alone, parasitism, in the form of one species taking advantage of the other's superior prey locating abilities, cannot be dismissed. While joint herding by both species was observed during the first encounter, foraging was clearly segregated during the second encounter, despite the apparent pursuit of the same prey species. Further data are required to determine the underlying factors behind these different foraging strategies.

A lower risk of predation via better predator detection and a decreased probability of any one individual to be attacked due to the larger aggregation size may also play a role (Norris and Schilt 1988). Fatal attacks by killer whales (*Orcinus orca*) on both false killer whales and bottlenose dolphins have been observed in New Zealand waters (Visser *et al.* 2010), showing that predation is a bona fide threat for both species.

The use of bubbles during prey capture has been widely described for mysticete species, in particular the humpback whale (*Megaptera novaeangliae*) (e.g., Sharpe and Dill 1997). Although using bubbles to herd prey during foraging is not common among delphinids, it has been observed in Atlantic spotted dolphin (*S. frontalis*) (Fertl and Würsig 1995), short-beaked common dolphin (*Delphinus delphis*) (Neumann and Orams 2003), dusky dolphin (*Lagenorhynchus obscurus*) (Trudelle 2010) and killer whale (Similä and Ugarte 1993). While bottlenose dolphins are known to use bubbles in a social context (Marten *et al.* 1996), there are very few records of this technique being used during feeding (Fertl and Wilson 1997). To date, bubble herding has not been documented for false killer whales. However, false killer whales have been observed to use bubbles underneath an observation vessel in Hawaiian waters where mahi mahi (*Coryphaena hippurus*) were known to be present. In this case, it was assumed that the bubbles were used to dislodge fish attempting to evade predation.²

The proximity of false killer whales to the surfacing bubbles during the first encounter suggests that they were producing at least some of the bubbles and observations during the second encounter confirmed that bottlenose dolphins were emitting bubbles. While coordinated foraging by mixed species groups of false killer whales and bottlenose dolphins has been observed previously in New Zealand waters (JRZ, unpublished data) the use of bubbles has not previously been recorded. Given that these are among the first observations of bubble herding for either species, it is not possible to ascertain if this constitutes broad scale behavior or a specific strategy adopted by this particular group or population.

Kahawai is a schooling coastal species endemic to areas within temperate Australasian waters (Paulin 1993). While kahawai is a known prey species for bottlenose dolphins in New Zealand waters (Constantine and Baker 1997), it has, to date, not been reported as a prey species for false killer whales. While predation on a coastal fish species may simply constitute opportunistic feeding, it may also suggest that foraging in nearshore or inshore waters may occur more frequently than is currently being reported.

Inshore movements of false killer whales are thought to be associated with inward flowing water masses and the pursuit of food sources (Kasuya 1971), which may explain their appearance in the coastal waters of the Hauraki Gulf. The scarcity of previous sighting reports in this region is consistent with the species preferring open oceanic waters (Odell and McClune 1999). The rare occurrence of false killer whales in the Hauraki Gulf, as well as photo-identification matches, suggests the two encounters observed over a five day interval likely represent the same group, or part thereof.

We recommend continued behavioral observations and photo-identification of false killer whales in New Zealand waters in order to gain insight into the nature and extent of these interspecific associations as well as population size. Given their infrequent occurrence and reported offshore distribution, satellite tagging (Baird

²Robin W. Baird, Cascadia Research Collective, 218^{1/2} West 4th Avenue, Olympia, WA 98501, June 2011.

et al. 2010) would be an advantageous method to gain further insight into habitat use, distribution and home range of this poorly described species.

ACKNOWLEDGMENTS

The authors thank the management and crew of MV *Dolphin Explorer* and Explore Images for their ongoing support including contributions of photographs and sightings data, as well as for providing a research platform. In particular, we thank Sonja Clemens, Sarah Gardner, Cathy Goeldner, Monica Greco, and Andy Light for supplying photographs and/or sighting information. We thank Robin Baird and three anonymous reviewers whose constructive feedback improved earlier drafts of this manuscript. Research onboard RV *Te Epiwhania* was funded by the New Zealand Department of Conservation, Auckland Conservancy, and Massey University. We thank Karen Elliott for assistance on the research vessel.

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Received: 13 December 2011

Accepted: 9 April 2012

Appendix D

ZAESCHMAR, J. R., I. N. VISSER, D. FERTL, S. L. DWYER, A. M. MEISSNER, J. HALLIDAY, J. BERGHAN, D. DONELLY and K. A. STOCKIN. 2014. Occurrence of false killer whales (*Pseudorca crassidens*) and their association with common bottlenose dolphins (*Tursiops truncatus*) off northeastern New Zealand. *Marine Mammal Science*. **30**: 596-608



Occurrence of false killer whales (*Pseudorca crassidens*) and their association with common bottlenose dolphins (*Tursiops truncatus*) off northeastern New Zealand

JOCHEN R. ZAESCHMAR,¹ Coastal-Marine Research Group, Institute of Natural and Mathematical Sciences, Massey University, Private Bag 102904, North Shore, Auckland 0745, New Zealand; INGRID N. VISSER, Orca Research Trust, PO Box 402043, Tutukaka 0153, New Zealand; DAGMAR FERTL, Ziphius EcoServices, 8112 Springmoss Drive, Plano, Texas 75025, U.S.A.; SARAH L. DWYER and ANNA M. MEISSNER, Coastal-Marine Research Group, Institute of Natural and Mathematical Sciences, Massey University, Private Bag 102904, North Shore, Auckland 0745, New Zealand; JOANNE HALLIDAY, 6 Kennedy Street, Opua 0200, Bay of Islands, New Zealand; JO BERGHAN, 4 Access Road, Kerikeri 0230, Bay of Islands, New Zealand; DAVID DONNELLY, Australian Marine Ecology, 82 Parsons Street, Kensington, Victoria 3031, Australia; KAREN A. STOCKIN, Coastal-Marine Research Group, Institute of Natural and Mathematical Sciences, Massey University, Private Bag 102904, North Shore, Auckland 0745, New Zealand.

ABSTRACT

On a global scale, false killer whales (*Pseudorca crassidens*) remain one of the lesser-known delphinids. The occurrence, site fidelity, association patterns, and presence/absence of foraging in waters off northeastern New Zealand are examined from records collected between 1995 and 2012. The species was rarely encountered; however, of the 61 distinctive, photo-identified individuals, 88.5% were resighted, with resightings up to 7 yr after initial identification, and movements as far as 650 km documented. Group sizes ranged from 20 to *ca.* 150. Results indicate that all individuals are linked in a single social network. Most observations were recorded in shallow (<100 m) nearshore waters. Occurrence in these continental shelf waters is likely seasonal, coinciding with the shoreward flooding of a warm current. During 91.5% of encounters, close interspecific associations with common bottlenose dolphins (*Tursiops truncatus*) were observed. Photo-identification reveals repeat inter- and intraspecific associations among individuals with 34.2% of common bottlenose dolphins resighted together with false killer whales over 1,832 d. While foraging was observed during 39.5% of mixed-species encounters, results suggest that social and antipredatory factors may also play a role in the formation of these mixed-species groups.

Key words: false killer whale, *Pseudorca crassidens*, bottlenose dolphin, *Tursiops truncatus*, mixed-species groups, interspecific associations, photo-identification, foraging, seasonality, New Zealand.

¹Corresponding author (e-mail: jzaeschmar@hotmail.com).

Very little is known of free-ranging false killer whales (*Pseudorca crassidens*) (Baird *et al.* 2008). Despite a worldwide distribution in tropical and warm temperate waters, this species is encountered infrequently in most locales where they occur (Odell and McClune 1999). Occurrence in most areas is primarily inferred from stranding records (Stacey *et al.* 1994), including for New Zealand (*e.g.*, Hector 1872, New Zealand Marine Mammal Stranding Database²). While mostly oceanic in distribution, false killer whales are known to venture close to shore at oceanic islands (Acevedo-Gutiérrez *et al.* 1997, Garrigue and Greaves 2001, Gannier 2002) and are observed occasionally in shallow continental shelf waters (Palmer *et al.* 2009). High site fidelity has been documented in some areas (Baird *et al.* 2008). Although cases of long-distance travel (Tomilin 1957, Leatherwood *et al.* 1989) and seasonal occurrence (Kasuya 1985) are reported, it is not clear if, or to what extent, populations of the species migrate.

Described as gregarious, false killer whales typically travel in groups of 20–100 animals (Baird 2008). Long-term associations between individuals have been recorded off Hawaii (Baird *et al.* 2008) and Costa Rica (Acevedo-Gutiérrez *et al.* 1997), with stable social clusters that differ in home range identified within the Hawaiian insular population (Baird *et al.* 2012). However, little is known about the species' overall social structure from other regions. There have been a number of mass and single strandings of false killer whales in New Zealand. Based on the number of stranded individuals, false killer whales rank second only to pilot whales (*Globicephala* spp.) within New Zealand waters (New Zealand Marine Mammal Stranding Database). Despite this prominent stranding history, few data or specimens have been collected from these events and data collected from free-ranging individuals are limited and primarily anecdotal (Gaskin 1972, Cawthorn 1981). The species is currently listed as *Not Threatened* within New Zealand waters, although confidence in the assessment is low due to poor data availability (Baker *et al.* 2010).

False killer whales have also been observed in nonaggressive associations with several other cetacean species (*e.g.*, Leatherwood *et al.* 1989), in particular with the common bottlenose dolphin (*Tursiops truncatus*) (*e.g.*, Best and Reeb 2010, Visser *et al.* 2010, Zaeschmar *et al.* 2013). While increased foraging success and predator detection and/or avoidance are suggested as possible factors in the formation of these mixed-species groups (Zaeschmar *et al.* 2013), the exact nature and extent of such interactions remain unclear.

The present study documents the occurrence and association patterns of false killer whales in the waters off northeastern New Zealand, taking into account seasonality, group size, and composition. Hypotheses for seasonality and population size and structure are provided. *Ad libitum* behavioral observations, focusing on the presence or absence of foraging are also discussed to further elucidate false killer whale use of the study area. Additionally, the study describes false killer whale interactions with common bottlenose dolphins, and provides hypotheses as to the nature and extent of such associations.

MATERIALS AND METHODS

All sightings were collected in five locations along an approximate 650 km stretch of the northeastern coast of New Zealand, between February 1995 and March 2012.

²New Zealand Marine Mammal Stranding Database, Museum of New Zealand Te Papa Tongarewa, % Department of Conservation, PO Box 10420 Wellington, New Zealand.

The study area is influenced by the shoreward progression of the warm, southeastward flowing East Auckland Current (EAUC) during December and its subsequent departure around May (Zeldis *et al.* 2004). The EAUC carries warm subtropical water (Sutton and Roemmich 2001) as well as associated marine fauna (Francis *et al.* 1999) into the study area. Sea surface temperature (SST) reaches 23°C during the austral summer and falls to 15°C in winter (Chiswell 1994). Records of false killer whales were primarily collected in waters off, or adjacent to, the Bay of Islands (BOI), (approximate position 35°S, 174°E), between February 1995 and March 2012. The area is characterized by a number of features; approximately 150 islands and islets with numerous bays and estuaries, and the Cape Brett peninsula, which intersects the warm EAUC, providing a large catchment area for nutrients (Baker and Madon 2007). Water depth between the islands and the mainland is generally <20 m while on the seaward side, water depth ranges from 50 to 120 m. The edge of the continental shelf is *ca.* 50 km to the northeast of Cape Brett.

Additional records of false killer whales were collated from the following four areas:

- (1) The Three Kings Islands (TKI) 2008, (approximate position 34°09'S, 172°08'E). Located 55 km northwest of New Zealand's North Island, situated near the continental shelf break, the area is characterized by submarine canyons and seamounts, with water depths generally ranging from 100 to >1,600 m.
- (2) The Poor Knights Islands (PKI) 2010, 2012, (approximate position 35°28'S, 174°44'E). Located 19 km off the New Zealand's northeastern coast, these small islands (271 ha) are volcanic remnants that rise steeply from the otherwise flat ocean floor. Water depth in the area ranges from *ca.* 80 to >150 m.
- (3) The Hauraki Gulf (HG) 2011, (approximate position 36°10'–37°10'S, 174°40'–175°30'E) is a shallow (<60 m), semi-enclosed body of temperate water. The seabed in the area is predominantly flat.
- (4) The Bay of Plenty (BOP) 2012, (approximate position 36°30'–38°10'S, 175°40'–178°00'E) is a large open embayment with a generally flat seabed, containing a small number of islands. The shoreward flow of the EAUC ceases around East Cape, the southern limit of BOP, with the current turning away from the coast (Stanton *et al.* 1997). Water depth in the area ranges from 50 to >200 m.

The majority of records (53.2%) were collected from the *Tutunui*, an 11 m fiberglass catamaran powered by twin 350 hp jet engines, with a cruising speed of approximately 30 km/h and an observer's eye height of 3 m above sea level. *Tutunui* is a commercial whale watching vessel staffed by experienced marine mammal observers that operates year round in BOI, although trips are more frequent between October and May. Additional records were collected from tour boats (*ca.* 11–22 m with 3–5 m observer eye height and travel speed of 15–35 km/h) and research vessels (*ca.* 5–6 m, observer eye height 2–3 m and survey speed of *ca.* 20 km/h) operating in the five study locations. Tour vessels encountered false killer whales opportunistically during wildlife/marine tours throughout the study area, while the research vessels encountered false killer whales during dedicated cetacean surveys. All vessels followed a similar, asystematic survey methodology which was dictated by factors such as prevailing weather conditions. Depth was determined by plotting the GPS coordinates of the sighting locations on the relevant bathymetric chart while distance from shore was measured by plotting sighting locations in ESRI ArcGIS version 9.3. Sea surface temperature (SST) was determined using onboard thermometers.

Ad libitum behavioral observations (Altmann 1974), focusing only on the presence or absence of foraging behaviors were recorded in transcript, audio log (dictaphone), or video form. Following Acevedo-Gutiérrez *et al.* (1997), foraging was defined by frequent, asynchronous dives with subgroups dispersing over larger areas, as well as by the presence of fish near the surface or sea birds feeding on fish remains near the whales. Prey species were confirmed by photographic record. As most of the observation platforms did not operate in the respective locations on a year round basis, seasonal occurrence was assessed from the records of *Tutunui*, which ran continuous trips up to twice daily in BOI between 1995 and 2007. Following Wiseman *et al.* (2011), a monthly index of false killer whale encounters was determined using a trip encounter rate (TER), which was calculated from the number of trips on which whales were encountered in proportion to the total number of trips undertaken that month. Results for each month from different years were pooled and calculated as an average sighting rate per 100 trips. To avoid pseudo-replication, only one sighting record per day was included in the analysis. A social network diagram of false killer whales photo-identified in the study area was produced using the program Netdraw 2.123.³ A spring embedded layout was chosen, placing more connected nodes at the center of the network while those with fewer connections were placed around the periphery. To test for the significance of encounter duration and presence/absence of foraging on group size estimates, a General Linear Model (GLM) with a negative binomial model approach was applied. All analyses were completed in R using the “Stats” (R Core Development Team⁴) and “Mass” (Venables and Ripley 2002) packages.

Following Baird *et al.* (2008) it was assumed that all false killer whales occurring in the area at any one time were part of the same group. This definition is supported by the infrequency of false killer whale encounters in the study area and the fact that false killer whales may at times disperse >20 km and still be moving in the same direction and engaged in the same behavior (Baird *et al.* 2008). Consequently, no spatial parameters were placed on group definitions. Given the frequency of association between false killer whales and bottlenose dolphins, for the purpose of this study, the term mixed-species group refers to associations between these two species only. Following Shane (1990), a mixed-species group was defined as any number of individuals of one species observed in apparent association with the other species and generally moving in the same direction and engaged in similar behavior (Fig. 1).

As the majority of observations described herein were of large, dispersed groups containing more than one cetacean species and recorded by several observers, group size estimates may be biased upwards. To address this possible bias, estimates were treated as pertaining to maximum group size, and in cases where minimum, best, and maximum group size estimates were available, the maximum estimate was used.

Two forms (coastal and oceanic) of the common bottlenose dolphin frequent New Zealand waters (Baker *et al.* 2010). The oceanic form is readily distinguishable based on gross morphology (Visser *et al.* 2010); they are comparatively more robust and typically exhibit wounds and scars, presumed to be inflicted by the cookie cutter shark (*Isistius* spp.) (Dwyer and Visser 2011). In contrast, the New Zealand coastal form does not usually exhibit cookie cutter shark scarring (Visser *et al.* 2010). The

³Borgatti, S. P., 2002. NetDraw software for network visualization. Analytic Technologies, Lexington, KY. Available at <http://analytictech.com/Netdraw>.

⁴R Core Development Team. 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.



Figure 1. A mixed-species group of false killer whales and common bottlenose dolphins. Bay of Islands, New Zealand, January 2007. Photo: David Hall.

form of common bottlenose dolphin observed in association with false killer whales was determined from photographs and/or video footage of the respective encounters.

Standard photo-identification methods (Würsig and Jefferson 1990, Baird *et al.* 2008) were applied to identify individuals (Table 1). A *capture* was defined as one or more useable images of an individual taken on an independent day. Primary identification features included notches on or adjacent to the dorsal fin and permanent distinguishing features such as dorsal fin disfigurement. Secondary features included scars as well as fresh subdermal wounds such as those presumed to be the result of cookie cutter shark bites. Only primary features were used to confirm matches, with secondary features used only as an aid to identification. Individuals, as well as images, were graded according to the likelihood of successful recapture and matching. The quality of each image was assessed by its focus, contrast and the angle of the fin relative to the frame and graded on a scale of 1 to 4 with 1 being *excellent*, 2 being *good*, 3 being *fair*, and 4 being *poor* (Table 1). The distinctiveness of each dorsal fin was graded on a scale of 1 to 4 with 1 being *very distinctive*, 2 being *distinctive*, 3 being *slightly distinctive*, and 4 being *not distinctive* (Table 1). Only *distinctive* and *very distinctive* individuals and images of *good* or *excellent* quality were included in the analysis. Each new image was carefully examined and all matches were confirmed by at least two experienced matchers. Successful photo-identifications were entered into the New Zealand False Killer Whale Identification Catalogue (NZFKWC; JRZ, unpublished data⁵), and New Zealand Oceanic Bottlenose Dolphin Identification Catalogue (NZOBDC; JRZ, unpublished data), respectively.

RESULTS

Forty-seven sightings of false killer whales were recorded between 1995 and 2012. The majority of observations (33 of 47; 70.2%) were made in BOI (2005, 2007, 2009, and 2010, with no false killer whales encountered in 2006, 2008, 2011, or 2012), with additional records from TKI (2008) (10.6%, $n = 5$), BOP (2009, 2012) (10.6%, $n = 5$), HG (2011) (4.2%, $n = 2$) and PKI (2010, 2011) (4.2%, $n = 2$). The majority

⁵Both catalogs are curated by the senior author.

Table 1. Grading system and assessment criteria applied for determining image quality and dorsal fin distinctiveness used for photo-identification. The quality of each image was assessed by its focus, contrast and angle of the fin relative to the frame. Only images of *good* or *excellent* quality of *distinctive* and *very distinctive* individuals were included in the analysis.

Image quality grading	Assessment criteria
1 (<i>excellent</i>)	All quality criteria are met: sharp focus with clear contrast and taken at an angle that allowed a clear profile of the dorsal fin's leading edge.
2 (<i>good</i>)	One of the quality criteria was compromised but the information content remained intact, allowing for the identification of <i>distinctive</i> and <i>very distinctive</i> individuals.
3 (<i>fair</i>)	Two or more quality criteria were compromised allowing only for identification of <i>very distinctive</i> individuals.
4 (<i>poor</i>)	One or more quality criteria were compromised to the point that successful identification of the individual was not possible.
Dorsal fin distinctiveness	Assessment criteria
1 (<i>very distinctive</i>)	Multiple notches including large notches and could be identified from photos of all quality categories except <i>poor</i> .
2 (<i>distinctive</i>)	Multiple notches and could be identified from <i>fair</i> , <i>good</i> and <i>excellent</i> photographs.
3 (<i>slightly distinctive</i>)	Few notches and could only be identified from <i>good</i> or <i>excellent</i> photographs.
4 (<i>not distinctive</i>)	Clean fins (<i>i.e.</i> , no notches or other permanent distinguishing features) or showed notches that could only be seen in <i>excellent</i> images within an encounter but unlikely between encounters.

of sightings comprised of mixed-species groups (91.5%, $n = 43$). Encounter duration ranged from 10 min to 3 h 45 min ($n = 47$, $\bar{x} = 68.9$, $SD = 49.2$). False killer whales were rarely encountered. Records collected aboard *Tutunui* in BOI show 29 sightings during 6,108 trips on 4,082 discrete days between 1995 and 2007, resulting in an overall TER of 0.47 encounters per 100 trips. Sightings only occurred during the austral summer (December–February) (TER = 0.37, $n = 8$) and autumn (March–May) (TER = 1.33, $n = 21$) with TER highest in March (TER = 2.04, $n = 12$) and April (TER = 1.98, $n = 10$) (Fig. 2). False killer whales were encountered in SST ranging between 18°C and 23°C ($n = 47$, $\bar{x} = 20.5^\circ\text{C}$, $SD = 1.3$) (Fig. 2). Bottom depth for the sightings ranged from 25 to 350 m ($n = 47$, $\bar{x} = 105.3$ m, $SD = 86.7$) with 63.8% of encounters (30 of 47) occurring in waters <100 m deep. Distance from shore ranged from 0.2 to 67.4 km ($n = 47$, $\bar{x} = 9.2$, $SD = 14.8$).

Mixed-species group sizes ranged from 28 to *ca.* 400 ($n = 43$, $\bar{x} = 120.4$, $SD = 64.6$). Within those mixed-species groups ($n = 43$), species-specific group size estimates were made during 76.6% of encounters (36 of 47), with group size for false killer whales ranging from 20 to 150 individuals ($\bar{x} = 46.7$, $SD = 28.5$). Group size estimates for false killer whales observed in single-species groups ranged from 20 to 50 individuals ($n = 4$, $\bar{x} = 35$, $SD = 12.9$).

Between 2005 and 2012, *good* or *excellent* images of false killer whale dorsal fins were obtained during 15 encounters (BOI = 7; BOP = 3; TKI = 2; HG = 2; PKI = 1).

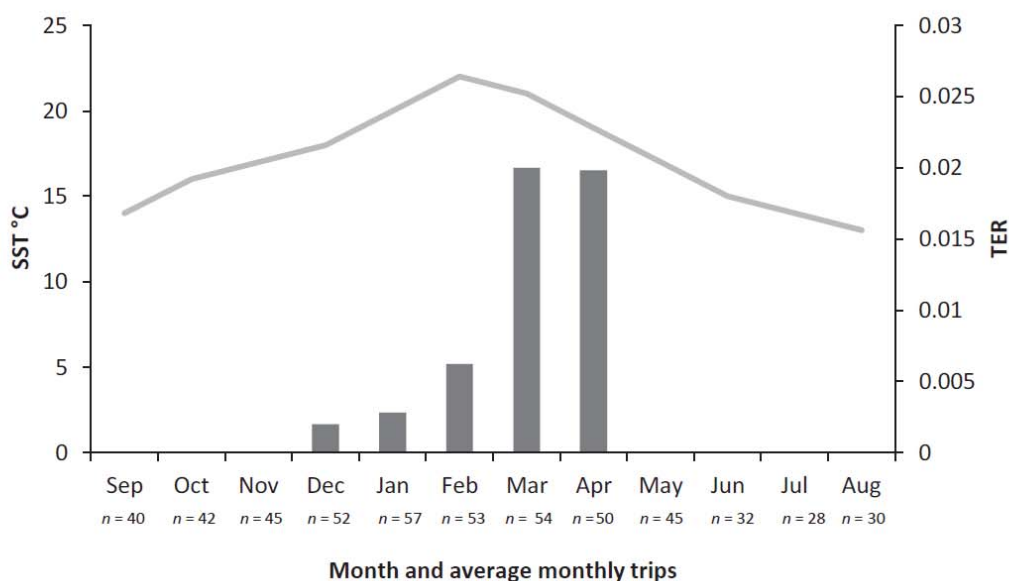


Figure 2. Mean monthly sea surface temperature (SST) and trip encounter rate (TER) of false killer whales in the Bay of Islands, January 1995–June 2007.

A total of 79 individuals were identified, 22.8% of which ($n = 18$) were considered *slightly distinctive* resulting in 61 *distinctive* or *very distinctive* individuals included in the analysis. Using only *good* and *excellent* quality photographs, 226 identifications of these 61 individuals were made ($\bar{x} = 3.7$, $SD = 2.1$). The number of individuals identified in each encounter ranged from 1 to 41 whales ($n = 15$, $\bar{x} = 14.7$, $SD = 11.1$), while the mean group size for encounters with identified individuals was 49 ($SD = 31.6$). Fifty-four individuals (88.5%) were resighted, with 70.5% ($n = 43$) encountered on three or more occasions and two individuals observed on eight occasions. Additionally, 85.2% ($n = 52$) were observed in more than one year and 62.2% ($n = 38$) were encountered in more than one of the five sighting locations within the study area. The longest time-frame between initial identification of an individual and its most recent resighting (disregarding sightings in between) was 2,551 d (approximately 7 yr, $n = 2$). The shortest time frame between any two resightings of an individual was 5 d ($n = 4$). Distances between sighting locations ranged from <1 km (BOI, $n = 29$, 1074 d) to *ca.* 650 km (TKI–BOP, $n = 8$, 284 d). Overall, the proportion of new identifications decreased as the rate of resightings increased (Fig. 3). While every encounter after the initial observation included previously cataloged individuals, previously unidentified individuals were only captured during 53.3 % of encounters ($n = 8$). The highest number of resightings between any two encounters was 29 individuals (January 2007 and December 2009, both BOI). All individuals photo-identified were linked by association in one large social network albeit separated into two social clusters (Fig. 4). Fresh subdermal wounds and scars presumed to be inflicted by cookie cutter sharks were observed on individuals during all encounters.

During 91.5% ($n = 43$) of encounters, false killer whales were observed in association with bottlenose dolphins with *good* or *excellent* quality photo-identification images obtained from nine encounters. The two species were considered mixed-species groups as individuals of both species were interspersed or in close association and generally acted as one homogenous group (Fig. 1). Based on the presence of cookie cutter shark scars on individuals, reported during all 43 encounters and

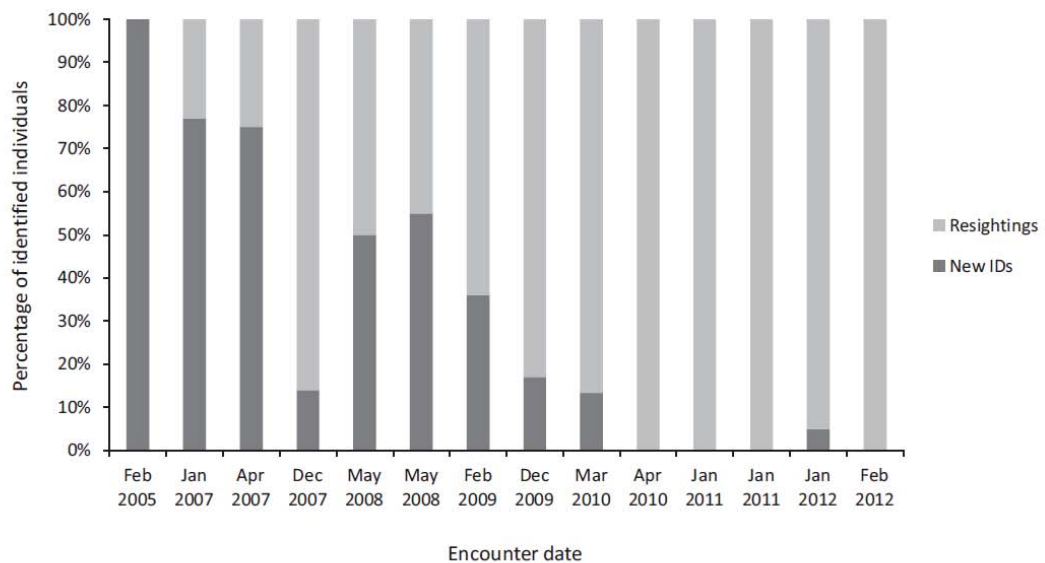


Figure 3. Percentages of newly (New IDs) and previously (Resightings) photo-identified false killer whales per encounter. Recorded off northeastern New Zealand between 2005 and 2012.

photographed during 15 encounters, bottlenose dolphins identified in association with false killer whales were presumed to be of the oceanic form. Group size for bottlenose dolphins were estimated during 36 encounters and ranged from 5 to *ca.* 250 individuals ($n = 36$, $\bar{x} = 62.8$, $SD = 42.8$). Using the same parameters as for false killer whales, 163 individuals were photo-identified, 8.6% ($n = 14$) of which were classified as *slightly distinctive* and not included in the analysis, resulting in 217 identifications of 149 *distinctive* or *very distinctive* individuals ($\bar{x} = 1.45$). Of these, 34.2% ($n = 51$) were resighted, with 10.1% ($n = 15$) observed on three or more occasions and two individuals (1.3%) on four occasions. Additionally, 28.2% ($n = 42$) were encountered in more than one year and 18.1% ($n = 27$) were observed in more than one of the five sighting locations within the study area. The longest time between first identification of an individual and its most recent resighting was 1,832 d (*ca.* 5 yr) documented for two individuals. Distances between sighting locations ranged from <1 km (BOI, $n = 14$, 1,074 d) to *ca.* 650 km (TKI-BOP, $n = 1,284$ d). The number of dolphins identified in each encounter ranged from 1 to 54 individuals ($n = 149$, $\bar{x} = 24.1$, $SD = 16.5$). Every encounter included previously photo-identified individuals. The highest number of resightings between any two encounters was 13 individuals. At least 34.2% of bottlenose dolphins ($n = 51$) had repeat associations with false killer whales involving 41 individuals (67.2%). Of these, 10.1% of bottlenose dolphins ($n = 15$) and 59% of false killer whales ($n = 36$) were re-encountered together on three or more occasions and during more than one year, with 8% of dolphins ($n = 12$) and 45.9% of whales ($n = 28$) observed together on three or more occasions as well as in different years and locations. Spatial separation/dispersion of individuals varied, ranging from all individuals being in close proximity (total area <1 km²) to wide distribution of individuals and subgroups within the whole mixed-species group (total area estimated at >5 km²). During all encounters, the two species were generally behaving as a single group, swimming within a body length of the other species (Fig. 1). However, clear segregation into conspecific subgroups within the primary group was also observed.

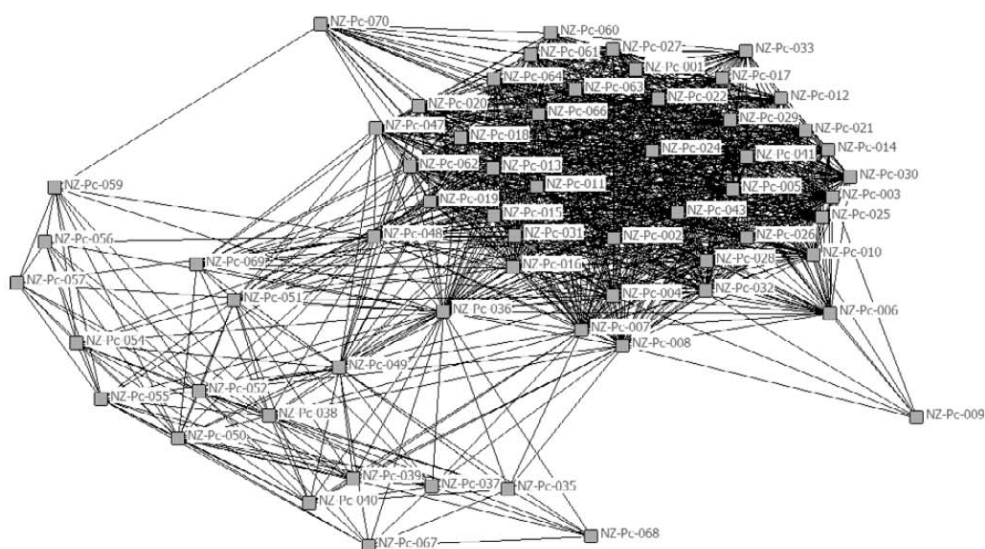


Figure 4. Social network diagram of 61 *distinctive* or *very distinctive* false killer whales photo-identified off northeastern New Zealand during 15 encounters between 2005 and 2012 using a spring embedded layout. Individual false killer whales with their corresponding catalog number are represented by nodes. Only individuals with *good* or *excellent* quality photos are included.

Note: Clustering suggests the existence of two social clusters.

Foraging was observed during 42.6% of all encounters ($n = 20$) and during three of the four single-species observations of false killer whales. Within the mixed-species observations ($n = 43$), foraging by at least one of the two species was observed during 39.5% ($n = 17$) of encounters. Mixed-species groups involved in foraging were generally larger ($n = 17$, $\bar{x} = 135.8$, $SD = 92.9$) than those groups where foraging was not observed ($n = 26$, $\bar{x} = 112.5$, $SD = 57.1$). However, encounters where foraging was observed were generally longer ($n = 20$, $\bar{x} = 82.5$, $SD = 55.4$) than those where it was not detected ($n = 27$, $\bar{x} = 58.9$, $SD = 42.3$). A negative binomial regression found no evidence that encounter duration ($P = 0.994$) and foraging behavior ($P = 0.420$) had any significant effect on group size estimates (Table 2). Evaluation of residuals found the assumption for homogeneous variance and normal distribution of residuals to be met and the P -value was considered reliable. Kahawai (*Arripis trutta*) was confirmed as a prey species for false killer whales and bottlenose dolphins on six occasions (BOI $n = 3$, HG $n = 2$, BOP $n = 1$). False killer whales were observed feeding on hapuku (*Polyprion oxygeneios*) during one mixed-species encounter (TKI).

DISCUSSION

Here we provide the first investigation of false killer whale occurrence in New Zealand waters. To our knowledge, only three other photo-identification studies have been conducted on the species worldwide: a dedicated, species-specific study currently examining distribution, social structure, home ranges, and genetic differentiation off Hawaii (Baird *et al.* 2010), a study resulting from cetacean surveys off Gabon and Côte d'Ivoire (Weir *et al.* 2013), and a previous study off Costa Rica (Acevedo-

Table 2. Negative binomial regression investigating potential predictors of false killer whale group size estimates. Model: $NB_{\text{Group Size Estimate}} = \alpha + \beta_{\text{Encounter Duration}} + \beta_{\text{Foraging Behavior}} + \varepsilon$.

Parameter	Estimate	Standard error	Z-value	Pr(> z)
Intercept	4.782000	0.135800	35.212	<0.001
Encounter duration	-0.000012	0.001593	-0.007	0.994
Foraging behavior	0.113030	0.161600	0.806	0.420
Null deviance: 45.216 on 42 degrees of freedom				
Residual deviance: 44.496 on 40 degrees of freedom				

Gutiérrez *et al.* 1997), which examined false killer whale occurrence as part of a broader three year study on dolphin ecology.

Despite their reported pelagic distribution (Wade and Gerrodette 1993, Baird 2008), false killer whales are known to approach close to shore at oceanic islands (Baird *et al.* 2010), with forays into shallow continental shelf waters also reported (Acevedo-Gutiérrez *et al.* 1997, Palmer *et al.* 2009, Zaeschmar *et al.* 2013). However, the latter are thought to be uncommon (Baird *et al.* 2008). Our data reveal repeat occurrence by individuals well within the continental shelf and in waters of less than 100 m in bottom depth. However, the extremely low encounter rate in the study area indicates that the species' distribution in New Zealand waters is likely also centered further offshore. The observed scars, presumed to be the result of cookie cutter shark bites, may provide further indication of false killer whale distribution in the area. Cookie cutter sharks typically occur in deep, tropical and temperate waters (Jones 1971, Jahn and Haedrich 1988) and their bite marks have been used as indicators to differentiate between cetacean populations (Dwyer and Visser 2011). The cookie cutter scars observed on both false killer whales and bottlenose dolphins suggest a potential distribution centered in warmer pelagic waters, which is consistent with the distribution generally reported for false killer whales (Baird 2008). Consequently, false killer whales encountered in the study area are likely to frequent deep oceanic waters as well as the shallow continental shelf region in which most of our observations were made.

While seasonality could only be assessed from records in BOI, all encounters from the different locations within the study area fall within the same period (December–May), further supporting the suggestion that occurrence in nearshore waters is seasonal. Changes in seasonal occurrence have been reported from Japanese waters (Kasuya 1971) and movements of false killer whales have been linked with warm water masses and migrations of prey (Tomilin 1957, Kasuya 1971, Leatherwood *et al.* 1989). Similarly, false killer whale occurrence in the study area coincides with the seasonal flow of the EAUC, a warm current that begins its shoreward progression towards northeastern New Zealand in December and completes its annual cycle by May (Zeldis *et al.* 2004). SST within the EAUC is approximately 2°C warmer than on the continental shelf (Sharples 1997) (Fig. 2). This current is associated with the arrival of warm water species such as various *Thunnus* and *Istiophoridae* species (Francis *et al.* 1999), the latter two being known prey items for false killer whales (Baird 2008). Our findings support the hypothesis that the seasonal occurrence of false killer whales in the study area may be the result of the species following the shoreward flow of the EAUC presumably in pursuit of prey, as has also been suggested for other regions (Tomilin 1957, Kasuya 1971, Leatherwood *et al.* 1989).

The observations of predation on kahawai, a schooling coastal species endemic to areas within temperate Australasian waters (Paulin 1993), has previously been

described in the study area (Zaeschmar *et al.* 2013), but have so far not been documented from other regions. Additionally, individual false killer whales observed to be feeding on coastal fish species were also observed to be feeding on hapuku in offshore waters. Predation on hapuku, a large demersal fish, inhabiting deep waters off the continental slope (Beentjes and Francis 1999), is more consistent with the feeding ecology reported for false killer whales from other regions (Odell and McClune 1999). These findings further support the suggestion that this population frequents both in- and offshore waters in New Zealand. The larger group sizes recorded during foraging observations may be the result of groups contracting when abundant prey is encountered. Indeed, satellite tagging in Hawaiian waters revealed that groups may disperse over more than 20 km (Baird *et al.* 2010).

All individuals photo-identified in the study area so far, are linked in a single social network and results suggest the existence of two social clusters within this social network, similar to the clustered social structure identified within the Hawaiian insular false killer whale population (Baird *et al.* 2012). However, given the likelihood that not all individuals were photo-identified in some of the encounters, the apparent clustering may result from incomplete capture of all individuals present in the groups observed. While the exact extent of this network remains unknown, it is certainly larger than the 79 identified individuals since nondistinctive individuals were present during all encounters, with maximum group size estimates during some encounters estimated to be as high as 150 individuals. These group size estimates are in sharp contrast to the actual number of individuals identified in each encounter. Possible explanations are that only certain individuals allow close enough approaches to observation vessels for successful capture or that group size estimates are biased upwards. Observer bias has to be considered when estimating the size of large mixed-species groups distributed over several square kilometers. The fact that group size estimates of false killer whales in single-species groups were considerably lower than those in mixed-species groups supports the likelihood of upward observer bias. However, false killer whale social networks can include hundreds of individuals, as is evidenced by mass stranding events, both in New Zealand (Baker 1981) and elsewhere (Ferreira 2008), suggesting that networks of this size also occur in New Zealand waters. It is worth noting that the rate of new identifications generally decreased over the course of the encounters described herein and that during some encounters no “new” individuals were identified at all. While false killer whales may initially appear to be infrequent visitors within New Zealand waters, photo-identification data presented here suggest recurrent use of the study area by a number of individuals and a level of site fidelity higher than generally assumed. Known individuals exhibited evidence of long-term associations, in many cases lasting years and spanning hundreds of kilometers. These findings are consistent with photo-identification from Hawaii (Baird *et al.* 2008) and Costa Rica (Acevedo-Gutiérrez *et al.* 1997). This, together with the high resighting rate over more than 7 yr, suggests a relatively small number of individuals with a reasonably high degree of site fidelity, albeit with a strong seasonal component, occurring in the study area. Further data are needed to ascertain if these individuals constitute a low-numbered, closed local population or form part of a larger, wide ranging metapopulation. Genetic sampling of the species in Hawaiian waters shows that individuals with high site fidelity form part of a closed population with a limited home range (Chivers *et al.* 2007, Baird *et al.* 2012).

The extremely high frequency of associations with bottlenose dolphins in the study area, including repeat associations between individual false killer whales and bottlenose dolphins spanning more than 5 yr and up to 650 km, suggests that such

associations are not random. False killer whales are known to associate with other delphinids and have been observed in close, nonaggressive association with a number of cetacean species (*e.g.*, Leatherwood *et al.* 1989, Baird *et al.* 2008), in particular the bottlenose dolphin (Mizue and Yoshida, 1961, Tsutsumi *et al.* 1961, Zhou *et al.* 1982, Leatherwood *et al.* 1989, Scott and Chivers 1990, Flores *et al.* 2003, Anderson 2005, Maze-Foley and Mullin 2006, Baird *et al.* 2008, Best and Reeb 2010, Zaeschmar *et al.* 2013). However, although bottlenose dolphin and false killer whale associations are reported previously in the literature and are considered relatively common (Reeves *et al.* 2002), very few studies have so far attempted to elucidate the possible extent and function of these associations (Zaeschmar *et al.* 2013). Associations (of any duration) between cetaceans are often linked to prey occurrence (Bearzi 2005). Indeed, apparent cooperation by false killer whales and bottlenose dolphins during foraging on large schools of prey has previously been reported within the study area (Zaeschmar *et al.* 2013) and has also been observed on several occasions during the present study. However, based on these observations alone, parasitism by one of the two species cannot be excluded. As foraging observations increased with encounter duration, it is likely that joint foraging is more prevalent than is currently being observed. However, the fact that foraging was not seen during many of the mixed-species encounters also suggests that additional factors may influence these associations. Standard group benefits such as increased predator detection and avoidance (Norris and Schilt 1988, Stensland *et al.* 2003, Kiszka *et al.* 2011) likely also apply. Indeed, predation by killer whales (*Orcinus orca*), for both false killer whales and bottlenose dolphins, is a *bona fide* risk in New Zealand waters (Visser 1999, Visser *et al.* 2010). Additionally, social factors may play a role and have been suggested as the driving factor behind some mixed-species associations (Bearzi 1997, Baraff and Asmutis-Silvia 1998). While results provide basis for the hypothesis that increased foraging success may be a contributing factor in the apparent stable nature of these associations, more data are required to ascertain the exact nature and function of their formation.

We have illustrated that false killer whales observed in coastal waters of northeastern New Zealand show long-term site fidelity to the region despite a strong seasonal peak in occurrence. All individuals identified so far are linked by association in one single social network, with repeat associations between individuals documented. Individuals were encountered in shallow coastal as well as deeper offshore waters, with occurrence close to shore possibly linked to the seasonal shoreward flooding of a warm ocean current. While further data are required to elucidate questions regarding population size and home range, a small local and possibly closed population cannot be ruled out. It is also evident that false killer whale associations with bottlenose dolphins are far from random in these waters. Our findings highlight the importance of long-term data collection and strongly support the need for dedicated research to be conducted on the species in New Zealand waters. Given these findings, a reassessment of the current conservation status in New Zealand may also be prudent.

ACKNOWLEDGMENTS

We would like to thank Great Sights Fullers Bay of Islands for allowing access to their historic whale watch records; the crew of the *Tutunui*, in particular Tammy Johnston; Rob Hunt from Explore Images, Sue Hews from Magic Memories, and Stuart Arnold and Rosie Roess from Dolphin Seafaris for access to many photographs; Elke Reufels (Department of Conserva-

tion) for permit assistance, and Emmanuelle Martinez and Jens Currie (Pacific Whale Foundation) for statistical input. The following people supplied additional photographs: John Allan, Alex Black, Robert Bradley, Tom Brough, Sonja Clemens, Mandy Everett, Sarah Gardner, Margaret Ginnelly, Cathy Goeldner, Monica Greco, David Hall, Tommy Hatwell, Lara Kay, Lily Kozmian-Ledward, Gemma Langsdale, Karin Neumann, Daniel Panek, Taira Peters, Richard Robinson, Jean Paul Rouen, and Nathan Turner. Robin Baird, Marta Guerra, and Tom Brough provided helpful comments on earlier versions of this manuscript.

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Received: 28 October 2012

Accepted: 13 July 2013