South Pacific Whale Research Consortium

Population status of Indo-Pacific bottlenose dolphins, *Tursiops aduncus*, in the Solomon Islands and assessment of live-capture sustainability

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EXECUTIVE SUMMARY

In 2009, a Memorandum of Understanding was developed and a collaborative project was initiated between the South Pacific Whale Research Consortium, the Solomon Islands Ministry of Fisheries and Marine Resources and the Solomon Islands Ministry of Environment, Climate Change, Disaster Management and Meteorology with the aim of providing scientific advice to help inform management decisions involving the removal of dolphins from wild populations in Solomon Islands. Here, we specifically investigate conservation issues related to the live-capture of Indo-Pacific bottlenose dolphins, *Tursiops aduncus*, combining demographic and genetic tools.

Molecular identification and population differentiation

Dolphin exporters allowed access to captive dolphins to collect sloughed skin samples for DNA analyses (n = 33) and photographs. Molecular analyses on these samples were used to confirm the taxonomic status of the dolphins targeted for live-capture and export as being *T. aduncus*. Regional population structure was also investigated using mitochondrial DNA and the analysis revealed that Solomon Islands *T. aduncus* are highly differentiated from neighbour populations of New Caledonia, Australia and China/Taiwan.

Small boat surveys

Small boat surveys (n_{TOT} = 62) were conducted in November 2009, November 2010 and July 2011. Total research effort was 5196 km in coastal waters of Guadalcanal, Florida Islands, Santa Isabel and Malaita, and 1930 km covered in offshore waters. Because of logistical constraints, we were only able to survey parts of Guadalcanal (34%), Santa Isabel (22%) and Malaita (58%) coastlines. However, the areas where captured of *T. aduncus* occurred in the past were well covered (i.e., north-west of Guadalcanal and north-west of Malaita). A total of 123 groups of marine mammals, representing nine different species were encountered during boat surveys. Biopsy samples were collected from 71 individuals of five species for the purposes of genetic analysis.

Groups of *T. aduncus* (n = 45) were the second most-commonly encountered species, after spinner dolphins, *Stenella longirostris* (n = 55). *T. aduncus* were typically found in coastal habitat, at an averaged distance of 0.39 nm from the coastline - they were never seen in waters deeper than 100 m. Rate of group encounters with *T. aduncus* varied according to islands (Kruskal-Wallis rank test $H = 8.744$, df = 3, $p < 0.05$), with highest group encounter at south Santa Isabel and lowest group encounter at Malaita. On the other hand, there was no difference according to surveys or seasons.

Photo-identification and site fidelity

Photographs were obtained from most groups encountered and a particular effort was made to document groups of *T. aduncus* (44 groups were photographed). Dorsal-fin photographs of *T. aduncus* identified 225 unique individuals in the wild. Twenty individuals were re-sighted within the same year while 46 individuals were re-sighted between different years. All resightings but one (Florida Islands to Guadalcanal) were found within study sites, indicating a high degree of site fidelity and suggesting a demographic partitioning between the study sites. Therefore, the four islands or group of islands appear to shelter distinct populations, most likely isolated demographically from each other.

A total of 28 captive dolphins were photographically identified during visits to holding facilities in 2009. Knowing that some were released in Guadalcanal and Florida Islands in 2010 (about 14 individuals), we look for matchings with dolphins photographed in the wild in 2010 and 2011. Only one of the 2009 captive animals was re-identified on the North Coast of Guadalcanal, in July 2011.

Abundance estimates

Given the evidence for localised populations of *T. aduncus,* abundance was estimated independently for each of the four study sites. We used closed-population models for the three annual sampling periods, and found that they all yield consistent results. After correcting the abundance estimates for the proportion of unmarked individuals in the population, we found that the survey areas at Guadalcanal, Florida Islands and Santa Isabel shelter small populations of *T. aduncus* of $N \sim 100$ for the former and $N \sim 300$ for the latter. Abundance estimates for the west coast of Malaita were considered to be less reliable because of insufficient data. However, population size around in this area is probably also in the low 100s. Summing of the four *T. aduncus* populations abundance estimates suggest a total abundance of around 700-1,300 dolphins in the overall survey area.

Potential Biological Removal and past depletion

Using the capture-recapture estimates of abundance, we calculated the Potential Biological Removal (PBR), as a management tool used to set anthropogenic removals. Given the evidence of local population structure, we assessed the sustainable level of removals for each distinct population using conservative PBR recovery value (Fr = 0.1). Doing so, the PBR for south west Guadalcanal and Florida Islands was less than one dolphin every five years and the PBR for south Santa Isabel and west Malaita was less than two dolphin every five years. Based on these calculations, the authorized export quota (50 dolphins per year) and the effective number of dolphins exported since 2003 (average 12 dolphins per year) are unsustainable if concentrated on one or few local populations, as it has been the case so far.

Management advice

Considering that most captures happened around Honiara, Guadalcanal, it is likely that the local population has been depleted since the beginning of the trade (potentially as much as half of the population was removed). It is also likely that the local population using the west coast of Malaita has been depleted, although it is harder to assess because of uncertainties in population abundance. In order to avoid depletion of *T. aduncus* in the long term, it is recommended to:

- Develop a new management procedure taking into account the PBR and past exploitation with the study areas.
- Prohibit new removals outside the study area without further biological assessment.
- Set future species-specific quotas based on captures rather than the number of export, because the last does not account for mortality during local captivity.
- Impose a complete capture ban on the Guadalcanal population until future monitoring shows an increase in abundance.
- Finally, establish a ‘DNA register’ for individual identification of all dolphins taken for trade, including those now held overseas.

- Future effort should include the unsurveyed coastline of the bigger islands in order to better understand distribution of local dolphin communities and provide management information at an island scale, with a priority to Guadalcanal and Malaita were live-capture has been taking place.

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INTRODUCTION

Top predators such as dolphins and other small cetaceans have a fundamental influence on the biological structure and function of marine communities (Heithaus et al. 2008). As 'keystone species', the consequences of their removal by hunting, capturing or incidental mortality can vary, but an increasing number of studies show evidence of large-scale deleterious cascading effects (e.g., Myers et al. 2007). They are also 'umbrella species' because conservation actions that mitigate threats to them are likely to improve the prospects for the protection of other organisms, as well as the ecosystem itself (Mann et al. 2000, Roberge & Angelstam 2004). A third aspect particularly relevant to the South Pacific, is that these charismatic megafauna play an important role in human culture. Indeed, many Pacific island cultures have myths, legends and traditional uses of cetaceans, indicating the importance of these species in the identities of people, their way of life and their heritage (SPREP 2008, Whimp 2008). For all these reasons, it is increasingly recognized that there is a need to improve their managment and conservation (Garibaldi & Turner 2004, Roberge & Angelstam 2004, Hoyt 2005), and in particular, it is a priority to assess and ensure the sustainability of any kind of removals through by-catch, direct kill or live-capture.

The Solomon Islands have a long history of hunting dolphins by driving groups onto beaches for slaughter (Dawbin 1966, Takekawa 1996). Several villages, especially on the island of Malaita, have been engaged in traditional drive-hunts for several decades or longer. The hunt provides teeth, which are used as dowry, traditional currency and adornments, and meat, which is consumed or sold locally (Dawbin 1966, Takekawa 1996, Reeves et al. 1999). In 2003, live-capture export trade was initiated, representing a new form of dolphin exploitation in the Solomon Islands. For this, dolphins are captured in the wild, held locally in captivity for training, then exported overseas for the purpose of public display, although some are kept in the Solomon Islands for display and breeding programs. Initial attempts to maintain captive dolphins were made with several species, including pantropical spotted dolphins (*Stenella attenuata*), spinner dolphins (*Stenella longirostris*) and Risso's dolphins (*Grampus griseus*). These were not successful, probably because of the difficulties to train and keep these species alive. Since these early efforts, the capture for export trade has concentrated on Indo-Pacific bottlenose dolphins, *Tursiops aduncus*, which are usually considered to adapt better to captivity, have better temperaments and are less susceptible to disease and stress than some other species of dolphins (Reeves et al. 1994). To date, the Indo-Pacific bottlenose dolphin is the only species that has been exported from the Solomon Islands overseas based on official export records.

According to Solomon Islands CITES Authorities, a total of 108 Indo-Pacific bottlenose dolphins were exported between 2003 and 2011 (UNEP-WCMC 2012). This represents the minimum number of dolphins that was removed from the wild since the beginning of the trade. Note, however, that the real number of dolphins removed is unclear and could be much larger than 108. Some individuals are still currently in captivity in Solomon Islands (and therefore not included in the total export number), and there is no official record of accidental deaths during capture, or deaths while in captivity. However, anecdotal accounts and media reports suggest that such loss happened multiple times (Parsons et al. 2010). Furthermore, it appears that, in 2010, a minimum of 14 dolphins were released from two facilities into the wild after an unknown period of captivity. The fate of these dolphins is unknown but release of captive animals into the wild has proven very difficult and unsuccessful elsewhere (Rose et al. 2009). The large majority of dolphins were captured around the capital Honiara, on the northwest coast of Guadalcanal. However, the last export of 25 dolphins in 2011 was composed of animals captured on northwest of Malaita, probably in 2010.

The Government of Solomon Islands currently permits up to 50 dolphins to be exported per year. However, based on the current state of knowledge of Indo-Pacific bottlenose dolphins throughout their range, international experts have suggested that this level of removal is unlikely to be sustainable (Reeves & Brownell Jr. 2009). Indeed, if an international standard ‘rule of thumb’ allowing 1% or 2% of a population to be removed annually was applied, the local *T. aduncus* population targeted by live-capture would have to be at least 2,500 or 5,000 to sustain the permitted level of exports. However, the species has a limited coastal range throughout much of the Indo-Pacific Ocean and, where studies have been conducted, the populations have generally been found to be small, a few hundred individuals at most (Wang & Yang 2009).

To investigate this issue, we developed a project that aims to improve knowledge of the population status of the targeted species and to assess the sustainability of live-capture for targeted local populations. Our primary objectives were: 1) to confirm the species identity of captured dolphins; 2) to describe the typical community structure of *T. aduncus* in the Solomon Islands; 3) to estimate the abundance of *T. aduncus* local populations, including in the area where most captures occurred; and 4) to calculate Potential Biological Removal within the study area as a tool for management of any future anthropogenic removals (Wade 1998).

Estimating cetacean abundance is not an easy task. Several types of survey methods can be used but some are not ideal or practical for this study. For instance, airplane line-transect studies can be useful but species identification of dolphins can be difficult from the air. It is particularly true in the case of Indo-Pacific bottlenose dolphins, since they cannot be easily differentiated from other species such as the common bottlenose dolphin, *T. truncatus*. Therefore, we choose to conduct small-vessel surveys for photo-identification work (i.e. individual recognition using unique markings on the dorsal fin) using mark-recapture analysis techniques. This method is the most appropriate technique in our case as the studied species is known to show a high level of individual distinctiveness in other areas near Solomon Islands (e.g. 70% in New Caledonia, Oremus et al. 2009), so the majority of the dolphins can be photo-identified. Photo-identification surveys also have the advantage of providing valuable data on population structure by investigating whether individual animals move between islands.

Once information is available on population structure and abundance, takes (removals) from marine mammal populations can be assessed in a variety of ways (see Reeves & Brownell (2009) for a review). Here, we decided to apply an internationally recognized method used to set limits for anthropogenic removals under the US Marine Mammal Protection Act and elsewhere, called “Potential Biological Removal” (PBR), which aims to ensure that human-caused removals are below levels that could lead to population depletion (Wade 1998). The PBR effectively sets take-limits where the mortalities occur at particular locations and times and are, at least in principle, directly observable (Lonergan 2011). The main advantage of

PBR is its simplicity: a value can be calculated from a single abundance estimate and without any direct estimation of population trends. The development of this method was a deliberate response to the difficulty of collecting data on the marine mammal populations (Taylor et al. 2006).

In this report, we first investigate the taxonomic status of captive dolphins using molecular techniques to confirm that they belong to the species *T. aduncus*. Indeed, species identification of captive dolphins so far was based on external morphological characteristics such as the total body length of adults, the shape of the beak or the presence of spots (Ross et al. 2003). However, the taxonomy of *Tursiops* sp. is rather complex and still not fully resolved, particularly in the Indo-Pacific Region. At least two species of *Tursiops* are thought to occur in the Solomon Islands (*T. aduncus* and *T. truncatus*), and recent studies also suggest the existence of a putative third species, described in nearby Australia (Charlton-Robb et al. 2011). Therefore, the question of the taxonomic status of the captured dolphins needed to be clarified. We then present results of a regional analysis of population genetic structure to investigate long-term connectivity between Solomon Islands *T. aduncus* and populations from neighbouring areas (e.g. New Caledonia).

Using photo-identification and capture-recapture techniques, we investigated *T. aduncus* individual movements and patterns of site fidelity between among four study sites presenting different islands or groups of islands. Such analyses can help to determine if *T. aduncus* found around different islands are isolated from their neighbours or belong to the same population, which has important implications in regards to the scale at which management decisions should be applied. We also searched for resightings of captive dolphins in the wild, knowing that a number of them were released at Guadalcanal and Florida Islands in 2010. Capture-recapture analyses were then used to provide population size estimates. Finally, the latter results served to assess the sustainability of dolphin removals for local populations, on the basis of the current quota of export as well as on the official number of dolphins exported since 2003.

MANAGEMENT BACKGROUND OF THE PROJECT

Despite a long history of traditional dolphin drive-hunts, until recently little attention has been given to marine mammal conservation or management in the Solomon Islands. However, in 2003, the development of a new enterprise of live-capture dolphin trade received considerable media coverage followed by numerous critisisms from wildlife activists, environmental agencies, and foreign governments. Concern was expressed by major intergovernmental groups, including CITES (Convention on International Trade in Endangered Species), CMS (Convention on Migratory Species) and IUCN (International Union for Conservation of Nature), about the potential conservation implications of dolphin removals in the Solomon Islands (Reeves & Brownell Jr. 2009). An assessment of dolphin removals has also been recognized as a priority under the SPREP (South Pacific Regional Environment Program) Whale and Dolphin Action Plan 2008-2012 and the CMS Pacific Cetacean Memorandum of Understanding (MoU).

The export of dolphins was banned by the Solomon Islands Government after the controversial shipment of 28 Indo-Pacific bottlenose dolphins to Mexico in 2003. However, the export ban was later challenged and overturned in court, and the live-capture and export trade resumed. Nevertheless, the Government decided to set a quota of 100 dolphins, of any species, to be exported per year, which was later reduced to 50 dolphins per year (UNEP-WCMC 2012). In August 2008, a workshop was held in Samoa by IUCN-CSG, focusing on the status and potential implications of *T. aduncus* removals from wild populations, with the Solomon Islands as a study case. Discussions focused on the status of *T. aduncus* populations and on how to conduct a research program that could provide decision makers with the robust data needed to help in management decisions involving the removal of dolphins from wild populations. This workshop was attented by dolphin experts from around the world, including four representatives from South Pacific Whale Research Consortium, SPWRC (M. Oremus, C. Garrigue, S. Taei and S. Childerhouse). It provided the opportunity to initiate communication between the SPWRC and representatives of the Solomon Islands Government (Mr. John Leqata and Mr. Joe Horokou). Following this initial contact, the Solomon Islands Government was invited to attend the next SPWRC annual meeting in February 2009 to further discuss dolphin removal issues and the potential for collaborative efforts that could take advantage of the SPWRC expertise in the assessment of cetacean populations' status (SPWRC 2009). This has resulted in the joint development of a research proposal between the SPWRC, the Solomon Islands' Ministry of Fisheries and Marine Resources (MFMR) and Ministry of Environment, Climate Change, Disaster Management and Meteorology (MECDM), taking into account recommendations made at the population assessment workshop organised by the IUCN in August 2008 (Reeves and Brownell 2008). The main aim of this collaborative effort was to overcome disagreements surrounding the use and associated conservation and management issues for dolphin populations in the Solomon Islands. This would be accomplished by developing an independent research project that would provide the Government with local scientific knowledge to help in management decisions involving the removal of dolphins.

In May 2009, a National Dolphin Technical Committee (NDTC) was formed by the Solomon Islands Government to ensure the finalisation and endorsement of a National Dolphin Management Plan of the Solomon Islands. The Committee is composed of relevant government agencies and NGOs. One of the key objectives of the committee is to oversee the development and implementation of biological surveys to investigate the status of dolphin populations in the waters of Solomon Islands. In June 2009, a researcher from the SPWRC (M. Oremus) travelled to the Solomon Islands to meet decision makers and to further discuss the research proposal written in February 2009 and the feasibility of such biological surveys. This resulted in the development of a MoU. The MoU was completed in November 2009 and signed in February 2010 by the SPWRC, the Solomon Islands MFMR and the Solomon Islands MECDM. The first biological surveys started in November 2009, followed by a second survey conducted in November 2010 and a final survey in July 2011. One researcher (M. Oremus) from the SPWRC has led the surveys, working in direct collaboration with officers from the MFMR and MECDM. An interim report was prepared in June 2011 (Oremus et al. 2011) while results from the present report were presented at a workshop held by the Solomon Islands Government, and facilitated by UNDP-SEMRICC, at Honiara in May 2012 .

METHODS

Dolphin facilities surveys

Contact was initiated with the dolphin exporters holding dolphins in captivity in the Solomon Islands during the course of our project in order to gather information on the species and number of individuals in pens. An attempt was also made to collect dorsal fin photographs of captive dolphins, as well as skin samples. For skin samples, the skin-swabbing technique (Harlin et al. 1999) was explained to the trainers so that they collect the samples themselves. This technique consists of using a sterilized nylon scrub pad that is swabbed on the dorsal or lateral surface of the dolphin to remove and retain epidermal cells. It has the advantage of being almost non-invasive but has the disadvantage of providing poorer quality genetic material (Harlin et al. 1999).

Field survey Effort

Study area

The Solomon Islands is an island nation located in Melanesia, Oceania (South West Pacific, 8°00’S and 159°00’W), between Papua New Guinea and Vanuatu. It consists of nearly 1,000 islands, representing over 5,000 km of coastline (Figure 1). The continental shelf around these islands is usually narrow, and the ocean floor quickly falls to several hundred meters depth. The Solomon Islands are part of the Coral Triangle which is recognized as the global centre of marine biodiversity (Allen 2008) and a global priority for conservation (Briggs 2005). The climate is typical of a tropical area being characterised by high and rather uniform temperature and humidity and, in most areas, abundant rainfall in all months (http://www.met.gov.sb/). East to southeast winds prevail from May to October, although not usually as strong as in other Pacific regions further south or east. West to northwest winds prevail from about November to April and are usually lighter than the southeast trades and much less persistent.

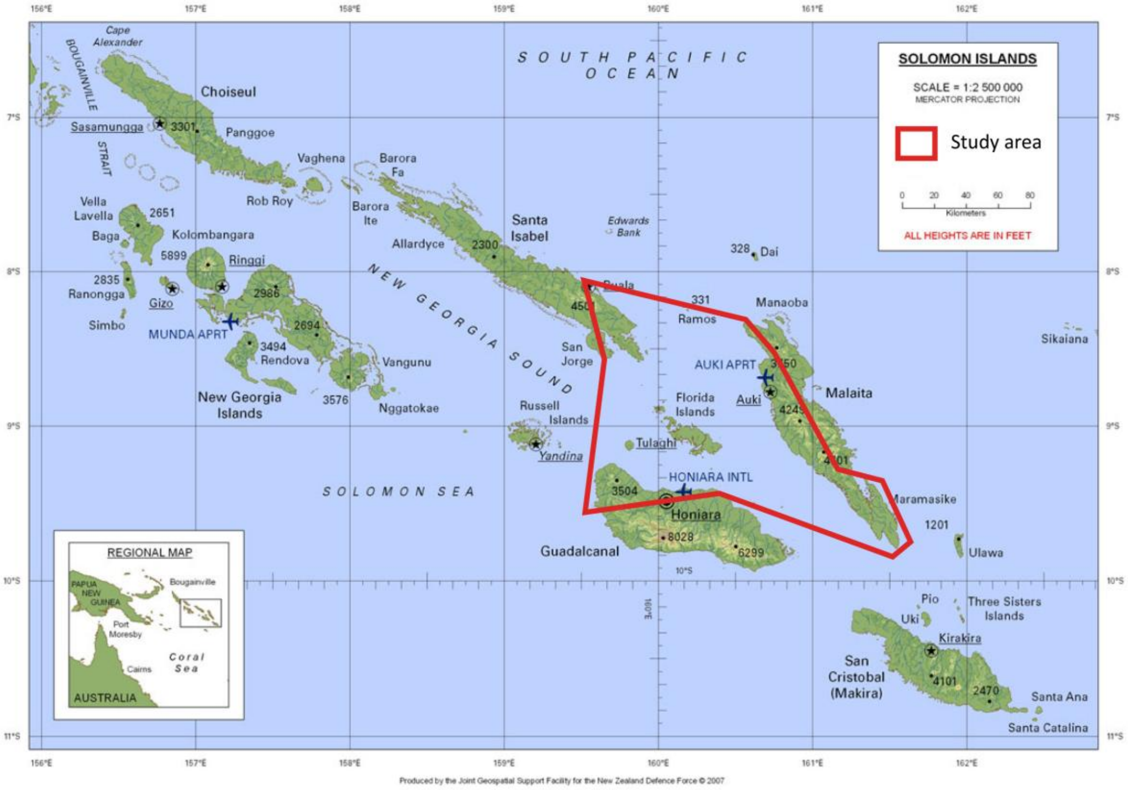

Figure 1: Map of the studied area in the Solomon Islands

Boat surveys

From November 2009 to July 2011, three series of small-boat surveys were conducted over one month each. In total, we conducted 62 surveys: 19 in November 2009, 20 in November 2010 and 23 in July 2011. The research vessel (6 m) was purchased by the MFMR specifically for this project (Figure 2).

In order to answer our research questions, we focus our effort at four islands or groups of islands of the eastern part of Solomon Islands: Santa Isabel, Malaita, Guadalcanal and the Florida Islands (Figure 1). Because of the large size of the islands and logistical constraints, it was not possible to cover the entire coastline of Guadalcanal, Santa Isabel and Malaita (Figure 1). However, the study area encompasses the locations were all captures of *T. aduncus* occurred, i.e., the north-west coast of Guadalcanal and the north-west coast of Malaita.

Figure 2: Research vessel for dolphin surveys in Solomon Islands

Effort was primarily concentrated within one nautical mile (nm) of shore, with survey lines roughly parallel to the coast. We choose this survey design because Indo-Pacific bottlenose dolphins appear to prefer near shore continental shelf waters and areas with rocky and coral reefs, sandy bottom, or sea grass beds (Reeves & Brownell Jr. 2009). They can be found in waters more than 200 m deep but are much more common in water less than 100 m deep (Wang & Yang 2009). Initial studies in the Solomon Islands by R.H. Defran have found this to be true for this area as well (Reeves and Brownell 2009). However, substantial search effort was also made offshore during our study, including multiple crossings between islands, to help confirm *T. aduncus* habitat preferences.

The research team was usually composed of a boat driver from MFMR, one or two photographers from MFMR and/or MECDM and one cetacean expert from SPWRC that recorded data on tape recorder and collected biopsy samples and photographs.

Data collection

We recorded the geographic positions of each group of marine mammals encountered during the surveys with a Global Positioning System (GPS). In this study, a “group” is defined as a spatial aggregation of animals that appears to be involved in a similar activity (e.g., foraging, socialising, resting or travelling, Shane et al. 1986). For each encounter, group size was estimated by visual counts, recording the minimum, maximum and best estimates.

Dorsal fin photographs were taken of as many individuals as possible, regardless of distinctive marks or vicinity to the boat, using digital SLR cameras (15 megapixels) equipped with telephoto-zoom lens. For each group, we measured the closest distance to shore by importing sighting locations into Google Earth and by using the ruler tool. Approximated depth for each sighting was estimated using the application “iSailor” in which sighting locations were placed onto official bathymetric maps of the studied area. The following maps were used: A4634, A3997, A1747B, A1747C, A1747F, A1750_1 and A1766B. Depth classes were defined as follow: less than 10 m; between 10 m and 20 m; between 20 m and 50 m; between 50 m and 100 m; more than 100 m.

The Paxarms© system was employed to collect small skin biopsy samples (Krützen et al. 2002). It uses a small biopsy dart fired from a modified 22-caliber veterinary rifle equipped with a variable pressure valve (Figure 3). This type of system was shown to have minimal impact on small cetaceans (Noren & Mocklin 2012, Tezanos Pinto & Baker 2012). Biopsies were only collected on individuals presumed to be adults based on large body size. Samples were preserved in 70% ethanol and stored at -20°C for subsequent analyses, at the University of Auckland, New Zealand.

Figure 3: Paxarms biopsy system

Laboratory procedures

Genetic samples processing

Total DNA was isolated from genetic samples (including skin swabbing from exporters) by digestion with proteinase K followed by a standard phenol: chloroform extraction method

(Sambrook et al. 1989) as modified by Baker et al. (1994) for small samples. A fragment of the 5' end of the mtDNA control region (d-loop) was amplified via PCR using the primers light-strand, tPro-whale M13-Dlp-1.5 (5'-TCACCCAAAGCTGRATTCTA-3', Dalebout et al. 1998), and heavy strand, Dlp-8G (5'-GGAGTACTATGTCCTGTAACCA-3', as reported in Dalebout et al. 2005). All amplification reactions were carried out in a total volume of 20 µl with 1 × Ampli-Taq buffer, 2.5 mM MgCl2, 0.4 µM of each primer, 0.2 mM deoxyribonucleotide triphosphate (dNTP) and 0.5 U of Ampli-Taq® DNA polymerase. The PCR temperature profile was as follows: a preliminary denaturing period of 2 min at 94°C followed by 35 cycles of denaturation for 30 s at 94°C, primer annealing for 45 s at 55°C and polymerase extension for 40 s at 72°C. A final extension period of 10 min at 72°C was included at the end of the cycles. Sex of DNA samples was identified by co-amplification of the male-specific *sry* gene and the ZFX positive control gene, as described by Gilson et al. (1998).

Photographs processing

Individuals within each group were identified using notches on the dorsal fin, shape of the dorsal fin, scarring and skin pigmentation. For every individual within each group, the best left- and right-side photographs were selected and graded for quality using four parameters: focus, exposure, orientation and percentage visible (Oremus 2008). For each criterion, the photographs were assigned a grade from 1 (bad) to 5 (excellent). The final quality score of each dorsal fin was calculated as the average grade over the four criteria. All photographs ranking 1 for at least one criterion were excluded from subsequent analyses, along with the dorsal fin images that rated less than 3.5 on average. Cut-off values were chosen after an overall comparison of the photographs according to their markings, to minimize the likelihood of mis-matches. Each individual represented by at least one photograph of sufficient quality was given a distinctiveness rating, based on marks on the dorsal fin visible from either left- or right-side (Oremus 2008). Rating was as followed: (1) not distinctive, (2) slightly distinctive, (3) distinctive, and (4) very distinctive. Dorsal fins photographs from captive dolphins were assessed for quality and distinctiveness using the same protocol.

Every individual showing distinctive mark(s) (rated (2) to (4)) were compared to each other, to identify re-sightings. A catalogue of unique individuals was created and re-sighting events were classified as “within” or “between” islands and “within” or “between” years.

Furthermore, we compared distinctive dolphins held in captivity to the catalogue of distinctive dolphins from the wild.

Analytical treatment

Species identity of captive dolphins

We compared the sequences of the mtDNA control region obtained from dolphins in captivity in the Solomon Islands to sequences published in GenBank and elsewhere from the two currently accepted species (*T. aduncus* and *T. truncatus*) and from the South-East Australia *Tursiops* population recently proposed as a new distinct species (*T. australis*, Charlton-Robb et al. 2011). Reference sequences are available from GenBank or from the authors (Appendix 1). For the purpose of this analysis, we used only sequences from animals sampled in the Indo-Pacific region. For *T. aduncus*, samples originated from China (Wang et al. 1999, Yang et al. 2005), East Australia (Möller & Beheregaray 2001, Möller et al. 2007, Wiszniewski et al. 2010), Hawaii (Martien et al. 2011), Indonesia (Wang et al. 1999), New Caledonia (Oremus et al. 2009) and South Africa (Natoli et al. 2008). For *T. truncatus*, samples came from China (Yang et al. 2005), East Australia (Charlton-Robb et al. 2011), French Polynesia (Tezanos-Pinto et al. 2009), Hawaii (Martien et al. 2011), Hong Kong/Taiwan (Wang et al. 1999), Japan (Kita et al. Unpublished), Kiribati (Tezanos-Pinto et al. 2009), New Caledonia (Oremus & Garrigue, unpublished), New Zealand (Tezanos-Pinto et al. 2009) and Palmyra Atoll (Martien et al. 2011). Sequences of *T. australis* are all from South-East Australia (Bilgmann et al. 2007, Charlton-Robb et al. 2011)

Sequences were aligned using the MUSCLE alignment method (Edgar 2004) as implemented in the software GENEIOUS (Drummond et al. 2009). The maximum length of the sequences available varied according to the different sources and therefore, for the purpose of our analyses, sequences were truncated so that they all represent the same portion of the gene. Variable sites and unique haplotypes for Solomon Islands sequences were identified and confirmed by visual inspection of peak heights using GENEIOUS.

The phylogenetic relationships of the mtDNA haplotypes were reconstructed using neighbour-joining (NJ) method, as implemented in MEGA (Tamura et al. 2011). Homologous sequences from two closely-related species, short-finned pilot whale (*Globicephala macrorhynchus*) and rough-toothed dolphin (*Steno bredanensis*) were used as outgroups. The robustness of phylogenetic groupings was assessed by bootstrap resampling (replicates: NJ, 5000; MP, 1000; ML, 200). Clades with bootstrap values > 70% were considered robust (Hillis & Bull 1993).

Genetic diversity and regional population structure

We further used mtDNA control region sequences to investigate the level of genetic diversity and connectivity between the population of *T. aduncus* in the Solomon Islands and the populations of the same species from three surrounding areas. We used only areas for which haplotype frequencies of samples were available and number of samples was large enough (>10 samples). These were: New Caledonia, East-Australia and China/Taiwan/Hong-Kong. Standard indices of genetic variation including nucleotide diversity and haplotype diversity were calculated using ARLEQUIN, v3.5 (Excoffier et al. 2005).

To test for genetic structure between geographic regions (Solomon Islands vs. New Caledonia vs. East-Australia vs. China/Taiwan/Hong-Kong), the exact test of population differentiation and analyses of molecular variance (AMOVA) were conducted as implemented in ARLEQUIN. For AMOVA, the differentiation was estimated using conventional F_{ST} (based on haplotype frequencies) and its nucleotide equivalent, Φ_{ST} (using Kimura 2-parameter), which incorporates information on the genetic distance between haplotypes. Significance was tested by 20,000 permutations of the original datasets. F_{ST} and Φ_{ST} are measures that indicate the extent of genetic differentiation among subpopulations and range from 0 (no differentiation) to 1 (complete differentiation).

Individual movements and site fidelity

We used capture-recapture histories based on photo-identification to investigate individual movements and site fidelity within the studied area. In the case of fidelity to a particular site, it is expected that individuals observed multiple times will be recaptured within the area where they were captured in the first place. If dolphins do not show fidelity to a particular site, then there is equal chance of it being recaptured off a neighboring island.

To provide a statistical assessment of site fidelity, inter-annual site fidelity was investigated using a maximum likelihood method assessing the probability, *pt*, that an individual observed in one particular area moves to another area between sampling periods. This method allows the number of identifications to be used as a measure of effort, allowing the inclusion of all years with any individual identification (Whitehead 2001). Then the probability that an individual remains in a study area one sampling period later is one minus the sum of the transition probabilities to the other areas. This was assessed with the program SOCPROG (Whitehead 2009). The option allowing for an external area was used to account for individuals that are not found in any of the four study sites.

Population abundance estimates

In light of the results obtained on site fidelity analyses (see Results section), we chose to estimate abundance for each study site separately (i.e., north-west Guadalcanal, Florida Islands, south Santa Isabel and west Malaita). Each of the three surveys (2009, 2010 and 2011) constitutes a sampling event for mark-recapture analyses with effort being broadly similar at each study site. There were 11 and 7 months between sampling periods 1 and 2, and 2 and 3, which has left sufficient time to allow animals redistribute themselves between sites and also for movements between islands if they occur.

For population estimation using mark-recapture methods there are two primary classes of models in wide use: closed models, in which it is assumed there are no additions to (birth, immigration) or losses (death, emigration) from the population of interest over the period of study (i.e., the closure assumption); and open models, that allow for additions or losses over the period of study. Open models are often more realistic for cetacean populations but they require long time sequence and sufficient data to adequately estimate parameters.

Therefore, since only three sampling periods were available here, with a relatively limited number of captures, we chose to run closed models. We note that across the 1.5 year period of the study, it is likely that some births and deaths occurred in the populations. However, we assumed that such effects were limited considering that the demographic parameters observed in other populations of the same species or closely related species indicate low calving and mortality rates. Additions and losses in the populations might also have happen through emigration and immigration but likelihood analyses suggest low rates of movements in the area (see Results section). Overall, the assumption of strict demographic closure is unlikely to be met here and this violation will result on overestimates of the true population sizes. On the other hand, mark-recapture abundance estimates can be underestimated as a result of heterogeneity of capture, which is the model assumption most-likely to be violated (Hammond 2001). It is important that such effect be considered; this source of variation can be taken into account by some closed population models that were considered here.

In order to select the best model of abundance estimate, we first attempted to use the Akaike's information criterion (AIC) model selection (Burnham & Anderson 2002), as implemented in program MARK (White & Burnham 1999). However, this procedure failed to provide satisfactory results for three of the four study sites, probably because of the limited number of captures available. Therefore, we choose to perform analyses of abundance estimates using four different models implemented in the program CAPTURE (Rexstad & Burnham 1991) and that we think are biologically most relevant to our study case. These are:

- Mo: is the simpler mark-recapture estimator which assumes that all individuals have the same probability of capture on each sampling occasion.
- Mt (Chao): is a model especially developed for sparse data where the probability of capturing the animal varied with time.
- Mh (Chao): is also a model developed for sparse data but where the probability of capture varied between individuals.
- Mth: accounts for both variation of capture probability in time and between individual.

As these estimates relied on natural markings to identify individuals, they refer exclusively to the population of marked animals, N. To include the unmarked portion of the populations and obtain estimates of the total populations, N_{total}, the proportion of unmarked individuals, θ, was computed on the basis of the proportion of unmarked dorsal fins estimated for each study site (Wilson et al. 1999). The variance of the total abundance estimate was estimated using the Delta method (Seber 1982, Wilson et al. 1999), where n is the number of animals from which θ was calculated:

$$\mathrm{var}(Ntotal)^n = N^2 total \left(\frac{var(N)}{N^2} + \frac{1-\theta}{n\theta}\right)$$

Confidence intervals for total population size were calculated by assuming that the error distribution was the same as for the estimate of the number of distinctive individuals (Wilson et al. 1999).

Assessment of sustainability

Over the various methods employed to date around the world, threshold values used to evaluate sustainability or acceptability of takes or removals of marine mammals range from 0.1% to 2% of a best estimate of abundance. Here, we choose to use the PBR method, which explicitly takes into account uncertainty and potential biases in the available information. A PBR is calculated using the following simple formula (Wade 1998):

$$\mathrm{PBR} = N_{min} \times 0.5\, R_{max} \times Fr$$

Where:

N_{min} = 20th percentile of the population size estimate,

R_{max} = Maximum annual population growth rate,

Fr = Recovery factor.

In the PBR, the 20th percentile of the abundance estimate (equivalent to the lower 80% confidence limit) is used to account for imprecision in the abundance estimates, as quantified in the CV. In regards to R_{max}, nearly all small cetaceans are thought to have rates of growth no higher than about 4% per year (R_{max}=0.04) (Wade 1998, 2002). Certainly no dolphin population has been observed to increase at a faster rate, and aspects of their life history (such as their relatively late age of sexual maturity and low birth rate) make faster rates unlikely for dolphins (Wade 2002; Reilly and Barlow 1986). Therefore, we used a default growth rate value of 0.04. A recovery factor (Fr) of 0.5 is standard for unexploited populations and was shown to be robust under many situations (Wade 1998), including when estimates of abundance R_{max} are potentially biased or when there are uncertainties about population structure. However, for very small or endangered populations, an Fr of 0.1 is recommended (Wade, 1998; Slooten & Dawson 2008). Indeed, a small population even without human-induced mortality is vulnerable in itself because of environmental and demographic stochasticity, and inbreeding (Lande 1999). Here, we conducted calculations using Fr = 0.1 and Fr = 0.5 for comparison, as minimum and maximum values recommended by the literature.

RESULTS

Captive dolphin holding facilities

In November 2009, 19 *T. aduncus* were held in captivity at the Honiara facility, Guadalcanal, while 27 were in the pens of the Gavutu Island facility, in Florida Islands. We were told that all of these dolphins were captured along the north coast of Guadalcanal, close to Honiara. In November 2010, a further visit to the Honiara facility showed that only eight dolphins were left, including six males and two females, according to the trainer. Three dolphins were exported in December 2009, while eight individuals were apparently released in front of the facility around July 2010. Also in November 2010, we found out that the Gavutu Island facility was closed with no dolphins left in the pens. Seven of the dolphins seen a year earlier were exported in December 2009. The fate of the remaining 20 dolphins is unknown. According to the former owner's blog (www.freethepod.com), six dolphins were released around June-July 2010 while the remaining 14 individuals died in captivity.

During November 2010 and July 2011, we also tried to visit a new facility belonging to a third entrepreneur and located at Mbungana, Florida Islands. Unfortunately, we were not granted access. Apparently, in November 2010, no dolphins were held in this facility. However, in July 2010, the owner confirmed that some dolphins were captured along the west coast of Malaita since our last visit and that they were currently held in pens at Mbungana. This was also directly confirmed to us by the fishing community at Taeloa, Malaita, which was in charge of capturing the dolphins. Although we had no clear evidence of how many dolphins were captured and from which species, 25 Indo-Pacific bottlenose dolphins were exported from the facility later in 2011 (UNEP-WCMC 2012). As of June 2012, an unknown number of dolphins are still held in this facility, as directly confirmed by the owner.

Tissue samples of captive dolphins were obtained from the facilities at Honiara (n = 16) and Gavutu Island (n = 17). Unfortunately, no samples could be obtained from a third facility at Mbungana Island, despite multiple requests to the owner.

Species identity of captive dolphins

DNA was extracted from all captive *T. aduncus* skin samples collected during this study. Molecular sexing on captive dolphins was successful for 14 samples, indicating a biased sex ratio of 12 males and 2 females (exact binomial test of goodness of fit, $p < 0.05$). The reason for the bias is unclear but it could be due to the selection process during capture event. A total of 16 sequences of the mtDNA control region were obtained from presumed Solomon Islands *T. aduncus* in captivity. These sequences were aligned with haplotypes of *Tursiops* sp. from other regions of the Indo-Pacific (n = 145) after being truncated to a fragment of 290 base pairs available for most sequences. Doing so, all sequences were compared for the exact same portion of the mtDNA control region gene. All phylogenetic reconstructions based on the consensus fragment show that sequences from captive dolphins in Solomon Islands cluster with haplotypes of *T. aduncus* from the Indo-Pacific region, with ≥ 70% bootstrap support (Figure 6). *T. aduncus* from South Africa and *T. australis* form separate monophyletic clades while *T. truncatus* were paraphyletic.

Genetic diversity and regional population structure

Despite representing the smallest sample sizes, the Solomon Islands and China/Taiwan/Hong-Kong *T. aduncus* showed a large number of haplotypes in comparison to East-Australia and New Caledonia (Table 1). This is further illustrated by higher haplotype diversity at the former two sites. The level of nucleotide diversity is particularly high for China/Taiwan/Hong-Kong but is also high in Solomon Islands, at least in comparison to East Australia and New Caledonia (Table 1).

Table 1: Summary of mtDNA genetic diversity for *T. aduncus* in Solomon Islands and neighbour populations

	N	# haplotypes	Haplotype diversity	Nucleotide diversity (%)
Solomon Islands	**16**	**7**	**0.8667 +/- 0.0567**	**0.9023 +/- 0.5744**
New Caledonia	79	2	0.5024 +/- 0.0134	0.3526 +/- 0.2658
East Australia	17	4	0.4950 +/- 0.0603	0.3855 +/- 0.2861
China/Taiwan/HK	43	9	0.9118 +/- 0.0424	1.7492 +/- 1.0013

We only found one shared haplotype between any of the four regions, which was haplotype 1 shared between East Australia and New Caledonia. Overall level of differentiation between the four regions was highly significant ($F_{ST} = 0.5489$, $p < 0.0001$; $\Phi_{ST} = 0.5491$, $p < 0.0001$). Exact tests of population differentiation show highly significant degree of population genetic structure between each region represented ($p < 0.0001$ for each comparison).

Field effort and data collection

Effort was broadly similar between the three series of small-boat surveys (Table 2), representing over 350 h of observation at sea for a total of 7126 kilometers (km) covered, including 5196 km of coastal effort and 1930 km of offshore effort (Table 2, Figure 4). The same areas were covered in 2009, 2010 and 2011 (Appendix 2), with the inclusion of the island of Savo during the 2010 and 2011 surveys. Because of the geographic proximity of the two islands, data from dolphin encounters at Savo were combined with Guadalcanal for analyses hereafter. Overall, weather conditions were good for the three series of surveys. Daily expeditions were only undertaken at Beaufort Sea State (BSS) less than four. Search effort was ended when BSS reached four but this occurred only rarely.

Table 2: Summary of research effort. Coastal coverage represents the percentage of coastline that was covered during the surveys for each of the four islands or group of islands.

SITE	Coastal coverage	# Surveys				Time on water (hours:minutes)				Coastal effort (km)			
		2009	2010	2011	**all**	2009	2010	2011	**all**	2009	2010	2011	**all**
Guadalcanal	34%	7	7	5	**19**	39:56	34:21	29:50	**104:07**	350	482	407	**1239**
Florida Islands	100%	3	5	7	**15**	18:20	37:33	36:44	**92:37**	215	496	550	**1261**
Santa Isabel	22%	5	4	6	**15**	26:15	25:05	27:50	**79:10**	383	335	511	**1230**
Malaita	58%	4	4	5	**13**	26:24	20:35	28:40	**75:39**	465	435	567	**1467**
Total	-	19	20	23	**62**	110:55	117:34	123:04	351:33	1413	1748	2035	**5197**

A total of 123 groups of marine mammals were encountered (Figure 5). These were represented by nine different species, including eight cetacean species and one sirenian (Appendix 3 and 4). Appendix 4 presents a summary of findings on species other than Indo-Pacific bottlenose dolphins. A total of 45 groups of *T. aduncus* were observed. They were the second most-commonly encountered species after spinner dolphins, *S. longirostris* (n = 55 groups).

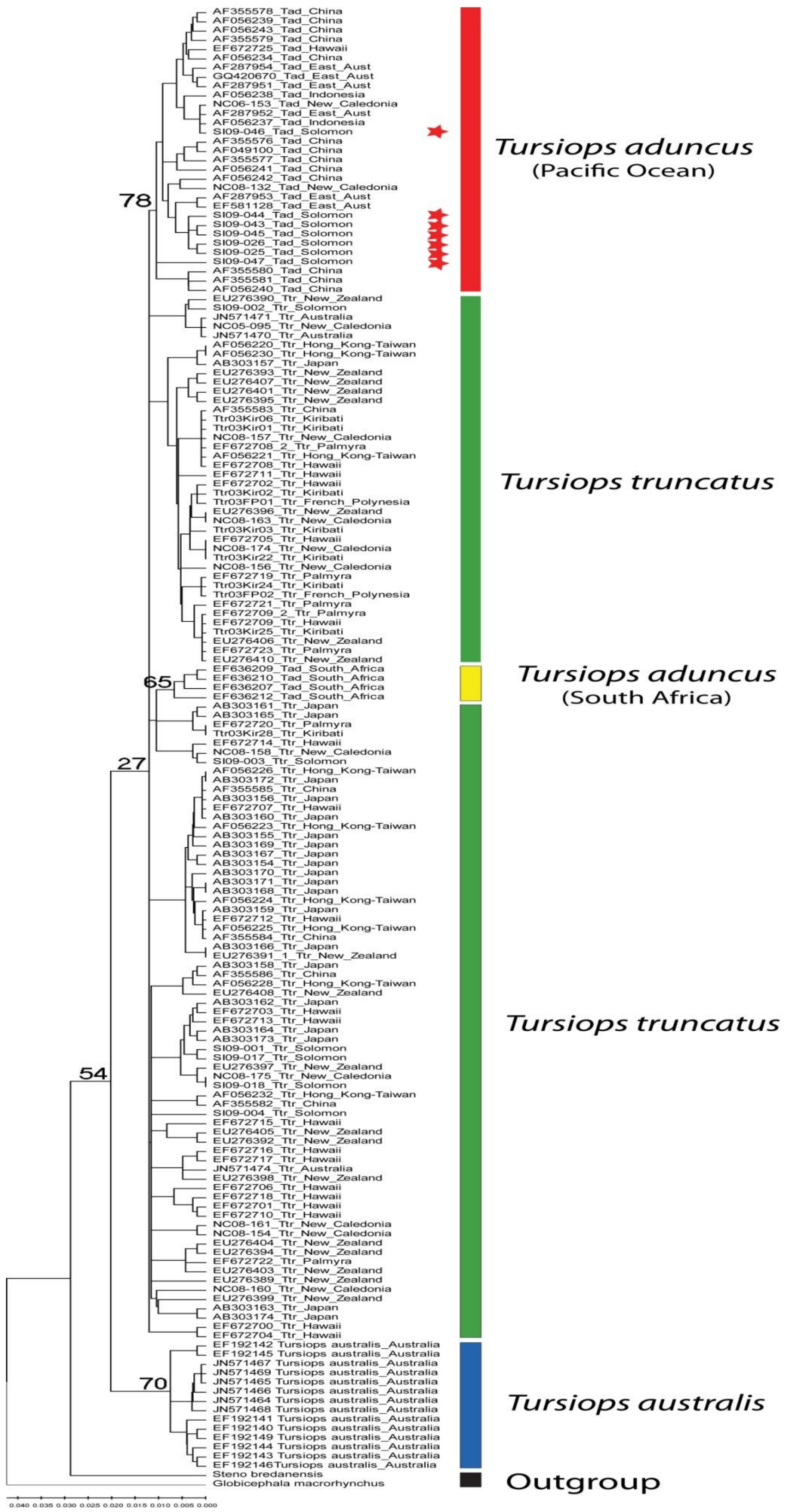

Figure 6: Phylogenetic relationships among mtDNA haplotypes from captive *Tursiops* spp. in Solomon Islands (red stars) and haplotypes from all species of *Tursiops* currently recognized, using the neighbor-joining method. Bootstrap values were enlarged at key nodes.

In each of the three years, 13 to 16 groups of *T. aduncus* were encountered, with a mean group size of 10.6 dolphins (SD = 10.5) but ranging from 1 to 60 individuals. *T. aduncus* was observed in mixed-species aggregations with *S. longirostris* (n = 11). In November 2009 at Guadalcanal, we observed a juvenile *S. longirostris* swimming along with a *T. aduncu*s on two instances, one day apart (Figure 8). On both encounters, the juvenile was the only *S. longirostris* in the group and on both encounters it was seen swimming with the same *T. aduncus*, as shown by distinctive marks on its dorsal fin. This particular *T. aduncus* was seen again in November 2010, with another Indo-Pacific bottlenose dolphin but mixed with a larger group of *S. longirostris*. The juvenile *S. longirostris* observed in November 2009 showed no distinctive marks and therefore, it is unknown if it was present again in this group. Unfortunately, we managed to collect only two biopsy samples of *T. aduncus* despite being the species with which we spent most time (total of 38h 57min).

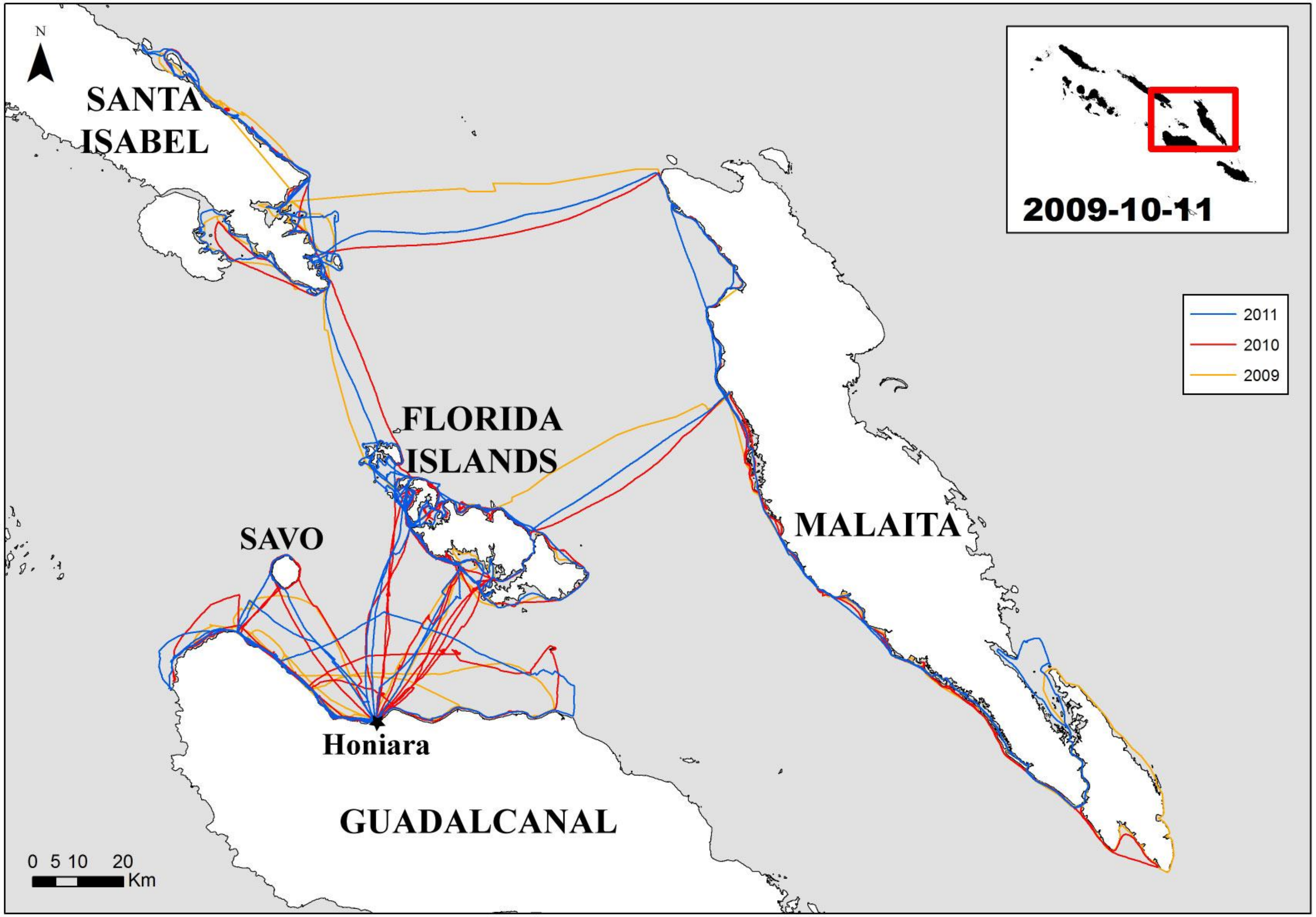

Figure 4: Survey coverage over the 3 years project.

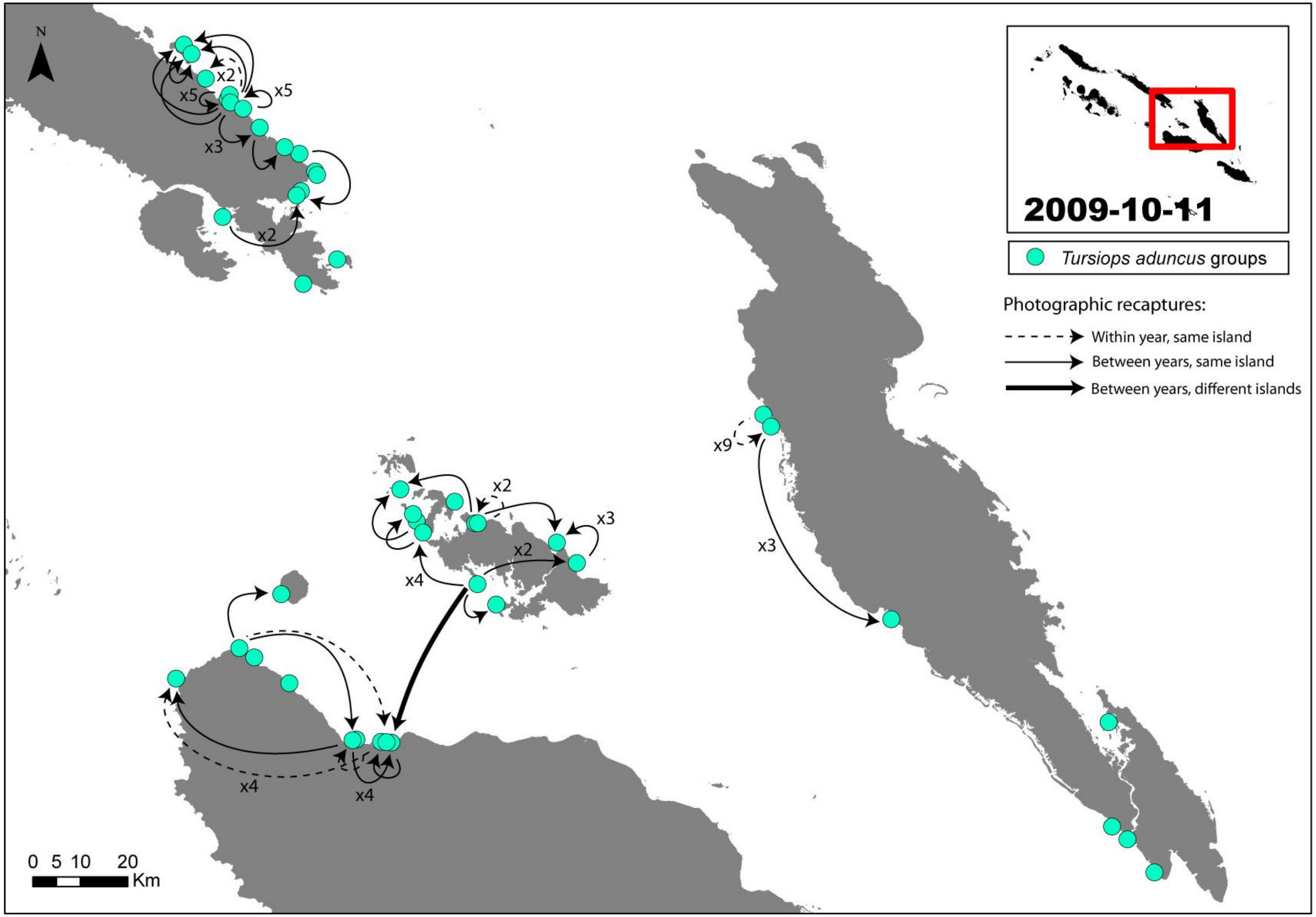

Figure 5: Geographic positions of *Tursiops aduncus* encounters in Solomon Islands during small boat surveys in 2009, 2010 and 2011, including individual movements within and between years as detected by photographic recaptures.

Habitat use

T. aduncus groups were encountered at an average of 0.39nm (SD = 0.26) from the coastline. In one occasion only, out of 45 encounters, a group was observed more than 1nm offshore (it was 1.5nm from coastline). The depth classes at which *T. aduncus* were encountered were distributed as follow: 5% at less than 10m; 26% at between 10m and 20m; 53% between 20m and 50m; 16% between 50m and 100m; 0% at more than 100m.

Encounter rates

T. aduncus encounter rates were computed for both groups and individuals. Overall, the average group encounter rate was 1.6 groups or 17 individuals per 100nm of effort within coastal waters. There was no significant difference in the rate of group encounters or in group size between any of the three surveyed periods (group encounters: Kruskal-Wallis rank test $H = 0.346$, df = 2, p = 0.841; group sizes: Kruskal-Wallis rank test $H = 2.361$, df = 2, p = 0.307). The highest rates of group and individual encounters were around the island of Santa Isabel (Table 4). The lowest rate of group encounters was found in Malaita but it was at Guadalcanal that we found the lowest rate of individual encounters. The rate of group encounters varied significantly between study sites (Kruskal-Wallis rank test $H = 8.744$, df = 3, $p < 0.05$) but not the rates of individual encounters (Kruskal-Wallis rank test $H = 5.821$, df = 3, $p = 0.121$).

Table 3: Summary of group and individual *Tursiops aduncus* encounter rates at four study sites in Solomon Islands

SITE	*T. aduncus* groups/100nm				*T. aduncus* individuals/100nm			
	2009	2010	2011	**all**	2009	2010	2011	**all**
Guadalcanal	1.1	1.9	1.4	**1.5**	4.2	8.1	19.5	**10.8**
Florida Islands	1.7	1.9	1.3	**1.6**	23.3	19.4	12.5	**17.0**
Santa Isabel	2.9	2.8	2.2	**2.6**	32.9	26.5	21.7	**26.5**
Malaita	1.2	0.4	1.0	**0.9**	11.2	3.0	23.9	**13.6**
Total	1.7	1.7	1.5	**1.6**	17.2	13.6	19.4	16.8

To investigate a potential seasonal effect on the occurrence of *T. aduncus* in Solomon Islands, we combined November 2009 and November 2010 data and ran comparative tests with data collected in July 2011. We found that there was no significant difference in the rates of group encounters (Kruskal-Wallis rank test H = 0.259, df = 1, p = 0.610) or the rates of individual encounters (Kruskal-Wallis rank test H = 0.259, df = 1, p = 0.610) between the two putative seasons.

Individual movements and site fidelity

Over 13,000 photographs were collected for the purpose of photo-identification analyses. Among these, 7,000 photographs were of *T. aduncus* and only this species is considered for the analyses below. *T. aduncus* photographs were obtained from all encounters but one. It represents 13 groups in 2009, 15 groups in 2010 and 16 groups in 2011. A total of 467 individuals were photographically identified but 34 were excluded from subsequent analyses because the photographs available for these dolphins were not of sufficient quality to be confidently matched with others. Of the remaining 433 identifications, we found that 293 had a distinctiveness rate of (2) to (3) and therefore, they present dorsal fin marks distinctive enough to be useable for photographic matching and capture-recapture analyses.

Table 4: Number of unique individuals identified and number of encounters per study site and per year, using photo-identification

	# unique individuals/# encounters			
SITE	2009	2010	2011	**All**
Guadalcanal	5/2	15/5	29/3	**36/10**
Florida Islands	16/2	35/5	14/3	**49/10**
Santa Isabel	46/6	32/5	39/6	**91/17**
Malaita	16/3	3/1	43/4	**50/8**
Total	83/13	85/16	125/16	**225/45**

The matching of the 293 distinctive dorsal fins revealed that 225 unique individuals were represented in the dataset, and therefore, we had 68 re-sighting events. The number of unique individuals identified at each of the four study sites range from 36 at Guadalcanal to 91 at Santa Isabel (Table 5). Twenty-two re-sightings were found within years (Table 6); seven at Guadalcanal, two at Florida Islands, four at Santa Isabel and nine at Malaita. Two of the Guadalcanal re-sightings were between the north coast of Guadalcanal and the island of Savo, 14 km to the north. There were no within-year re-sightings between any of the different study sites. Similarly, a total of 46 resighting events were found between years (Table 6). All of them are within the same study sites except one individual that was first observed in November 2009 around Florida Islands and was re-sighted in July 2011 on the north coast of Guadalcanal.

Table 5: Summary of overall re-sighting history between November 2009, November 2010 and July 2011 (within year/between years)

	Guadalcanal	Florida Islands	Santa Isabel	Malaita	All
Guadalcanal	**7/6**	-	-	-	-
Florida Islands	0/1	**3/15**	-	-	-
Santa Isabel	0/0	0/0	**3/22**	-	-
Malaita	0/0	0/0	0/0	**8/3**	**-**
All	-	-	-	-	**22/46**

Likelihood analysis using SOCPROG shows that estimates of movement rates between study sites are all small (*pt* < 0.05) and, consequently, that probabilities for individuals to be re-sighted at the same site each year are high for all our study locations (*pt* > 0.7; Table 7). This is with the exception of higher probability of movement from Guadalcanal to Florida Islands (*pt* = 0.14), a value that remains low in comparison to the probabilities of being re-sighted at the same site.

Table 6: Probability of resighting at the same study site (diagonal) and probability of movement from one area to another, as estimated using likelihood method in program SOCPROG. Standard errors are shown in brackets.

		To Area:				
		Florida	Guadalcanal	Isabel	Malaita	*External area*
From Area:	Florida	**0.7598**	0.0226 (0.03552)	0.0427 (0.02600)	0.0544 (0.03240)	0.1205 (0.03132)
	Guadalcanal	0.1410 (0.02750)	**0.7071**	0.0139 (0.02860)	0.0408 (0.03229)	0.0971 (0.03568)
	Isabel	0.0001 (0.00844)	0.0255 (0.01683)	**0.9070**	0.0378 (0.03338)	0.0295 (0.03424)
	Malaita	0.0001 (0.01480)	0.0315 (0.02349)	0.0040 (0.02335)	**0.9297**	0.0348 (0.02987)
	External area	0.0937 (0.03671)	0.0273 (0.02965)	0.0866 (0.03683)	0.1020 (0.03278)	0.6904

Photo-identification of captured and released dolphins

Visits to dolphin holding facilities allowed identification of 14 distinctive dolphins each at Honiara and Gavutu in November 2009. In November 2010, we want back to the Honiara facility and identified four distinctive dolphins, including two individuals that were already identified in the same facility in 2009. Overall, 30 unique dolphins with distinctive marks on their dorsal fin were identified in holding facilities. Comparison of the captive dolphins' catalogue to the catalogue of dolphins from the wild has revealed only one match of a dolphin initially photographed in the Honiara facility in November 2009 and re-sighted on the North Coast of Guadalcanal in July 2011. None from Gavutu were resighted.

Population abundance estimates

Given the evidence of high site fidelity at the four study sites, abundances of *T. aduncus* were estimated separately for each site. To assess total population sizes, we adjusted all estimates for the portion of the populations not distinctive enough to be used for photo-identification. These represent 32% of individuals overall based on good quality photographs. Slight differences in this proportion were observed between each of the four study sites with the percentage of unmarked individuals ranging from 29% at Santa Isabel to 35% at Florida Islands and Malaita. At Guadalcanal, the proportion of unmarked individuals was assessed at 32% of all dolphins.

As expected, abundance estimates were slightly different depending on the model used (Table 8). Mh consistently provided the highest estimates of all models, while Mt resulted on the lowest estimates. Despite these differences, we found that regardless of the model used, estimates for one site are largely consistent with one another, with the exception of west Malaita. Indeed, estimates for this area are less consistent showing substantial differences between models. Furthermore, they show very large CV and wide CI in comparison to the other sites, indicating that best estimates for that island are not precise.

We found that for north-east Guadalcanal, Florida Islands and south Santa Isabel, the total population sizes were in the low hundreds whatever the model used (Table 8). Note, however, that south Santa Isabel appears to shelter a larger population than north-east Guadalcanal and Florida Islands. In regards to west Malaita, the Mt model shows the smallest, but still large, CV and suggests a population of similar size than in Santa Isabel while the models Mo, Mh and Mth suggest that the population could be larger. However, as stated above, large CV and wide CI, indicate that best estimates for that area should be interpreted cautiously. We calculated global abundance over the study area by summing the best estimates for each of the four study sites. In this area, we found that the total population of *T. aduncus* probably numbers 750-1300 individuals.

Table 7: Summary of abundance estimates at the different study sites using four closed-population models, including coefficient of variation (CV) and confidence interval (CI). Estimates are shown after adjustments for the proportion of un-marked individuals.

	Mo				Mt (Chao)				Mh (Chao)				Mth			
	N	CV	95% CI low	95% CI high	**N**	CV	95% CI low	95% CI high	**N**	CV	95% CI low	95% CI high	**N**	CV	95% CI low	95% CI high
north-west Guadalcanal	**132**	0.33	84	264	**98**	0.28	69	182	**162**	0.39	93	359	**126**	0.43	75	175
Florida Islands	**131**	0.20	102	198	**120**	0.18	95	177	**158**	0.25	111	271	**137**	0.35	92	309
south Santa Isabel	**252**	0.16	197	352	**249**	0.18	191	361	**327**	0.22	232	510	**287**	0.31	187	567
west Malaita	**459**	0.53	200	1263	**283**	0.44	148	685	**644**	0.58	256	1888	**570**	0.60	223	1751
Global estimate	973	0.25	582	2078	750	0.18	503	1405	1291	0.30	691	3027	1120	0.32	577	2802

Assessment of Potential Biological Removals

Values of PBR were calculated for each of the four study sites using the abundance estimates and CVs obtained through the various models (Table 9). PBR values are very consistent for each of the study sites regardless of the model used to estimate population abundance. Only PBR values for west Malaita were substantially larger when using the most optimistic abundance estimates. Using the conservative recovery value recommended for populations subjected to past exploitation or very small populations (Fr = 0.1), all levels of sustainable removals are less than 1 individual captured per year, and as low as 1 individual removed every five years for north west Guadalcanal and Florida Islands. The PBR value was less than one dolphin removed every two years for south of Santa Isabel.

Table 8: Summary of PBRs obtained at the different study sites and overall, depending on the model of population abundance and on two values of recovery factor (Fr= 0.1 or 0.5)

	Mo		Mt		Mh		Mth	
	PBR (Fr=0.1)	PBR (Fr=0.5)	**PBR (Fr=0.1)**	PBR (Fr=0.5)	**PBR (Fr=0.1)**	PBR (Fr=0.5)	**PBR (Fr=0.1)**	PBR (Fr=0.5)
north-west Guadalcanal	**0.2**	1.0	**0.2**	0.8	**0.2**	1.2	**0.2**	0.9
Florida Islands	**0.2**	1.1	**0.2**	1.0	**0.3**	1.3	**0.2**	1.0
south Santa Isabel	**0.4**	2.2	**0.4**	2.1	**0.5**	2.7	**0.4**	2.2
west Malaita	**0.6**	3.0	**0.4**	2.0	**0.8**	4.1	**0.7**	3.6
SUM of ALL	**1.5**	7.3	**1.2**	5.9	**1.9**	9.3	**1.5**	7.7

DISCUSSION

Species identification and population differentiation of live-captured dolphins

Molecular identification conducted on skin samples from captive dolphins confirm that the species targeted for live-capture around the Solomon Islands are Indo-Pacific bottlenose dolphins (Figure 6). This is not a surprise since it has always been assumed that it was the species of choice for export traders in the Solomon Islands. However, several studies have highlighted the complexity of *Tursiops* taxonomic status in regions surrounding the Solomon Islands, including the recent description of a presumed previously unrecognized species in south-east Australia (Charlton-Robb et al. 2011).

Regional population structure analyses revealed that, at the mitochondrial level, Solomon Islands *T. aduncus* populations are highly differentiated from neighbouring populations of New Caledonia, China/Taiwan/Hong-Kong and East Australia. None of the haplotypes described in Solomon Islands were found elsewhere, suggesting little or no female-mediated gene flow between these regions on a generational timescale. The limited connectivity suggested by mtDNA at a regional level highlights the risk of impacting local populations, which might have a low resilience due to absence of re-colonisation through neighbour areas. However, additional analyses would be needed, including nuclear markers (to examine male-mediated gene flow) and samples from Vanuatu and Papua New Guinea. The relatively high level of mtDNA diversity of Solomon Islands *T. aduncus* in comparison to New Caledonia and East Australia is positive indicator given the problems associated with low genetic diversity (Frankham et al. 2002). However, this high diversity should not be interpreted as a function of population abundance. From an evolutionary history point of view, lower levels of diversity in New Caledonia and East Australia than in Solomon Islands are not unexpected since the former two are located at the distributional limits of the species (Wang & Yang 2009).

Distribution and site fidelity

As expected, our results confirm that in the Solomon Islands, *T. aduncus* are preferentially distributed near coast (< 1nm) and in shallow areas, as usually described in populations of this species elsewhere (Wang & Yang 2009). Indeed, no group was observed further than 1.5nm from the coast or in waters deeper than 100m despite almost 30% of our effort being conducted offshore.

In places where the species has been studied extensively, the Indo-Pacific bottlenose dolphins appear to exhibit strong year-round residency (e.g., around Mikura Island, Kogi et al. 2004). Some seasonal movements may occur in some places, as suggested on the east coast of South Africa (Peddemors 1999), but it is unclear whether or not it involves the same species than in the Solomon Islands (see above). Here, comparison of November surveys to the survey in July does not suggest a seasonal variation in the occurrence of *T. aduncus* in the Solomon Islands. On the other hand, the levels of encounter rates (groups and individuals) suggest some differences of density between study sites with higher levels of encounter in the south of Santa Isabel. The reasons for this pattern are unclear but they could be due to variation in the amount of suitable habitat between the study sites as well as to the recent impact of live-capture on the populations of Guadalcanal and Malaita, which show the lowest levels of individual and group encounter rates.

Analyses of photographic recapture over the whole study area show that Indo-Pacific bottlenose dolphins in the Solomon Islands have a high level of site fidelity in comparison to interchange between study sites. Indeed, all but one of the 46 recaptures between years were made within one of the four study sites, suggesting that each of the study sites shelters a distinct population or community of *T. aduncus*. Although the detection of one movement from Florida Islands to Guadalcanal (about 40 km apart) shows that low levels of exchange are possible, the populations around each of these islands should be considered demographically independent when management and conservation decisions are to be taken. Indeed, if human-caused mortality occurs in only one area while the quota was based on total abundance over several populations, there is a risk of depletion and potentially extirpation within the area being impacted (Barlow et al. 1995).

Photographic capture-recaptures indicated broad movements within each study site (Figure 5). Therefore, it is likely that the boundaries of the populations of interest for this study extend beyond the survey areas, at least for Guadalcanal, Santa Isabel and Malaita. Supporting this assumption are photographic evidences from previous work by R.H. Defran that show movements of *T. aduncus* between the eastern point of Guadalcanal and Honiara, over 100 km away. On the other hand, we cannot discount the possibility that several distinct populations inhabit each of these islands. Indeed, recent studies on *T. aduncus* have shown clear evidence of small-scale population structure along the coast of Australia and New Caledonia (Oremus et al. 2009, Wiszniewski et al. 2010). Following such pattern, it is possible that several populations share the coast line of large islands such Guadalcanal, Santa Isabel and Malaita. Defining exact population range for the groups that were encountered during this study will require further investigations with extended surveyed areas.

Population abundance and survey design

All population abundance estimates suggest that three of the four study sites shelter populations in the low hundreds, at most (about 100 to 300 individuals). The only exception was west Malaita for which some of the estimates were above 500 individuals (Table 8). This could be explained by the fact that a larger area was cover around that island (Figure 4). However, abundance estimates for this area should be interpreted cautiously since they have large associated CVs (in particular with Mh and Mth models), indicating low precision in the results. The best estimate for west Malaita using Mt is less than half the estimate using Mh, suggesting a population size of less than 300. It is therefore difficult to decide on a reliable estimate for that area. The lack of precision for population estimates at west Malaita is in part explained by a very limited number of identifications in November 2010, and thus a low recapture rate. It is also possible that our estimates are biased upward by the capture of more than 25 dolphins between surveys 2 and 3, in July 2011. Considering that the number is relatively large in comparison to the assumed population size (in the low hundreds), these removals will violate the assumption of demographic closure in the closed population models.

For the other three study sites, all four models provide consistent estimates with reasonably small CVs,. A pattern of small resident populations was described for *T. aduncus* in other areas such as Amakusa-Shimoshima and Mikura Islands in Japan (Shirakihara et al. 2002, Kogi et al. 2004), Moreton Bay in Australia (Chilvers & Corkeron 2003) and New Caledonia (Oremus et al. 2009). Therefore, our results are consistent with previous knowledge on this species.

However, it is important to note that our abundance estimates only provide an assessment of the number of individuals that used the surveyed areas during the course of our study. These assessments cannot be considered to reflect the total population sizes for each of the four islands or group of islands that were visited, with the exception of the Florida Islands, which where we surveyed 100% of the coastline. The coastline of Guadalcanal, Santa Isabel and Malaita were only covered partially (Table 2). Given the survey limits, it is possible that the dolphin population(s) around these islands encompass more individuals that we could assess. Rough adjustments of total population sizes at Guadalcanal, Santa Isabel and Malaita could be calculated by correcting our estimates by the percentage of coastline covered. However, such adjustments would be simplistic and could be misleading if used to define or assess quota of removals. For instance, it is likely that each of these islands is home to more than one population of Indo-Pacific bottlenose dolphins. If so, it would be inappropriate to set a quota of capture over an entire island. Furthermore, R.H. Defran’s study shows that the range of *T. aduncus* observed near Honiara is likely to extend as far as the eastern point of Guadalcanal. The abundance estimate provided here for north-west Guadalcanal is thus likely to encompass a population that use more than 34% of the island coastline. Thus, these adjustments are only provided here to give a rough idea of the likely implications of current authorized removals at global scale. Further surveys are needed before such adjustments could be used with confidence to set catch allocations.

Sustainability of past removals

One of the primary goals of this project was to assess the sustainability of the currently authorized quota of export – this was set at 50 dolphins per year as of 2009 (UNEP-WCMC 2012). Although this quota does not specify the species, it was known that only *T. aduncus* are effectively targeted for captures and captivity. Therefore, even though the assessment made here only concerns one species it can be appropriately compared to the official quota. Although the quota presumably applies to all of the Solomon Islands, it is known that most of the captures are concentrated around Honiara. For this reason, our surveys were designed to assess the potential for depletion to populations in the regions with the highest likelihood of capture. Our effort was substantial and the entire area where captures occurred to date (north-west of Guadalcanal and west coast of Malaita) was covered with the addition of two other sites not directly impacted by this practice (Florida Islands and south of Santa Isabel). Although it would be desirable to survey cetacean diversity and abundance on a national scale, this would require survey effort that very few, if any, countries have been able to undertake.

The PBR method was chosen to assess the sustainability of *T. aduncus* removals from wild populations in the Solomon Islands because we consider it the most robust management procedure, given the available data. PBR was developed to account for the uncertainty inevitably associated with estimates of abundance over a limited time frame (Wade 1998). Indeed, management methods relying on detection of population trends have proven inadequate for marine mammals. The variance typically associated with population estimates of dolphins means that the statistical power to detect declines is low, and in many cases even a decline of 50% would not be detectable statistically (Taylor et al. 2006). Therefore, the time required to detect population declines (e.g. by aerial or boat surveys) is so long that management actions based on such detection would not be initiated until populations have been seriously depleted.

A first assessment of sustainability can be made by comparing the export quota of 50 dolphins per year to PBR values calculated from the abundance estimates of the current surveys. Over the whole study area (i.e., the four sites), it shows that even with the most optimistic estimate of sustainable removal based on high recovery factor (Fr = 0.5) and abundance estimate model (Mh model), the PBR is still far below the current authorized level of removal (maximum of 9.3 dolphins removed per year, Table 8). Therefore, it is likely that maintaining the current export quota, which does not include individuals that die in captivity before export, could lead to a decline and possible extirpation of the local populations of *T. aduncus*. However, the current quota applies to the whole of the Solomon Islands, of which only a portion of the coastline was surveyed. Therefore, it might be reasonable to adjust this PBR upward to account for unsurveyed areas, i.e., approximately 80% of the coastline. Doing so, the PBR could raise up to 46.5 dolphins per year which is still under the current authorized quota. However, we caution against such a simplistic assumption until further adjusted estimates are available for the entire country.

We note that the total number of dolphins exported (N = 108) since the beginning of the trade in 2003 is well below the current authorized annual quota when considering the period of nine years over which these exports occurred. From 2003 to 2012, an average of 12 dolphins per year was exported. Although this number could be considered sustainable, if distributed across populations throughout the Solomon Islands, it is unlikely to be sustainable on the local level of past collection. Most of the dolphin captured for export came from the north-west coast of Guadalcanal (n = 83), where the most optimistic PBR is 1.2 dolphins per year (Table 8). Even a rough adjustment of PBR values for the entire island of Guadalcanal (i.e. 34% coastline covered) would result on a maximum of 3.5 dolphins removed per year.

When putting past removals into perspective with the current abundance estimates for that Guadalcanal (Table 8), it seems likely that a large portion of the resident population was removed because of live-captures. It is also important to note that 83 exported dolphins represent only a minimum estimate of the dolphins removed from that population. Four dolphins remain in captivity in Honiara and there is little doubt that many dolphins died during capture or captivity before export (Parsons et al. 2010). Furthermore, some of the captured dolphins were subsequently released back into the wild after an unknown period of time. The fate of these individuals is unknown, with the single exception of the dolphin that was resighted in the wild after being initially photo-identified at the Honiara dolphin facility. Six dolphins captured in Guadalcanal were reportedly released in the waters of Florida Islands (i.e. outside their original habitat) after they were held in captivity at the Gavutu facility. None of the individuals that were photographically identified at Gavutu were resighted in the wild. All in all, the true number of dolphins removed from Guadalcanal (including mortality) certainly exceeds the minimum of 83 individuals. Assessment of capture impact in Malaita is more difficult because of uncertainties regarding the abundance estimate and thus, sustainable level of removal. However, we know that more than 25 dolphins were captured so far. Such a large number is unlikely to be sustainable.

Future management

The discussion above indicates that the current authorized dolphin export quota is likely to be unsustainable and that the local *T. aduncus* populations of Guadalcanal and possibly Malaita have been depleted by live-capture activities.

Our study indicates that several populations, or stocks, should be considered for management purpose in the studied area. Therefore, it is inappropriate to apply an overall quota that could lead to the depletion of one conservation unit where most of the capture would be concentrated (which is what happened so far). For this reason, we estimated PBR for each of the four sites where we identified distinct communities of dolphins. Based on photographic evidence suggesting demographic segregation between study sites, we chose to use four closed population models to estimate abundance. Slight differences were observed between models, but when applied to PBR estimates, we could see that recommended levels of removals were consistent across the models used. The only noticeable (but still small) difference that we found was for Malaita for which our abundance estimates were not precise and highly variable depending on the model applied. However, because this population has already been impacted and because the larger abundance estimates are characterised by the largest CV, it is recommended that the most conservative abundance estimates are retained for management measures (i.e. Mt model).

The parameters chosen to calculate PBR can vary accordingly with the studied species and characteristics of the population. Here, we used a conventional Rmax = 0.04 as typically reported for cetacean species, but it should be noted that slower maximum growth rates (< 0.02) were estimated for dolphin populations elsewhere, based on specific assessment for these populations (Slooten & Dawson 2008). Therefore, a maximum growth rate of 4% is potentially optimistic and could inflate PBR estimates. We choose to report PBR estimates calculated using both high and low recovery factors (Fr = 0.5 and 0.1. respectively) to provide a clear assessment of the sustainability of the current authorized quota and effective number of *T. aduncus* removals under a broad range of assumption. However, it has been recognized that PBR is not sufficiently precautious for small populations that involves additional considerations. For this reason, a recovery factor of 0.1 is recommended for management decisions regarding endangered species and populations (Wade 1998). Here, we can see that the three populations with reliable abundance estimates (Guadalcanal, Florida Islands and Santa Isabel) fulfil the IUCN Red List criteria D for endangered population due very small number of mature individuals (less than 250). A recovery factor Fr = 0.1 should therefore be considered for management decisions regarding live-capture of *T. aduncus* in the Solomon Islands.

Overall, the assessment of *T. aduncus* population status and sustainability of removals indicate that to prevent decline and ensure the persistence of local dolphin populations in the future, no more than one dolphin every five years should be captured from the areas of north-west Guadalcanal and Florida Islands, while no more than one dolphin every two and half years should be captured from the areas of south Santa Isabel and west Malaita.

Future survey effort and genetic samples from captive dolphins

As noted above, our surveys were designed to cover the areas of known distributions and past captures of *T. aduncus*, primarily the north-west of Guadalcanal and west of Malaita. However, it was not possible to cover the entire coastline of Guadalcanal, Malaita or Santa Isabel, and we cannot discount the possible existence of other distinct populations of *T. aduncus* outside the surveyed ranges. In regards to Guadalcanal, we note that R.H. Defran reported the movements of a substantial number of dolphins between Marau (eastern point of Guadalcanal) and Honiara, suggesting a home range extending across most of the North Coast of Guadalcanal (Reeves & Brownell Jr. 2009). In the future, it would be important to extend the range of our surveys in order to cover the entire coastline of these islands. This could provide improved management information at an island scale. Such surveys should be conducted in priority around Guadalcanal and Malaita were live-capture has been taking place.

We note that a substantial number of *T. aduncus* photo-identification data were collected between 2005 and 2009, primarily around Guadalcanal but also in the Florida Islands and Malaita, during a research project conducted by R.H. Defran. Considering that our study area largely overlaps with this of Defran's project, it would be particularly interesting to compare catalogues between the two studies. A large number of dolphins were captured around Guadalcanal during the 2005-2009 period and, therefore, a comparison could help assess with more accuracy the impact of live-capture on the local population.

Finally, the difficulty in collecting biopsy samples of free-ranging *T. aduncus* (only two samples were collected over the whole study period) prevented us from conducting an analysis of population genetic structure at a local scale. A number of samples could be obtained from captive dolphins but they all originated from Guadalcanal and, therefore, we could not assess population differentiation between islands. To overcome this problem, we attempted to get samples of the dolphins recently captured around Malaita, which would have provided an opportunity to compare *T. aduncus* populations from Guadalcanal and Malaita. This was not possible and the dolphins have since been exported overseas. However, it remains possible to conduct such work assuming that the new owners of the dolphins agree to provide small tissue or blood samples. A list of specific recommendations regarding the future management of *T. aduncus* live-capture is provided below. Further recommendations regarding traditional drive-hunt and potential for dolphin-watching tourism are also provided in Appendix 5.

RECOMMENDATIONS ON LIVE CAPTURE OF *T. ADUNCUS*

- Any quota to be set should be specific to the unit to conserve, i.e., the species and the population. Because the results of the study show that there are distinct populations of *T. aduncus* on the study site and because live-capture trade only targets *T. aduncus*, the quota should not be applied for the whole of Solomon Islands but for each of the distinct populations as identified by scientific studies.
- Given evidence of a likely past impact on the local populations of *T. aduncus* targeted for live-capture (namely Guadalcanal and Malaita), no future capture should be allowed in areas where data are unavailable on population status.
- Any future quotas should not exceed the calculated Potential Biological Removal (PBR) based on available estimates of abundance. Given past exploitation, we recommend use of the conservative recovery factor (Fr = 0.1).
- *T. aduncus* should not be taken during traditional drive-hunt since it was shown that this species typically forms small coastal populations that could not sustain the large number of catches usually taken by drive-hunters.
- Any quota should refer to “capture” event and not “export”, as it is currently the case. By referring to “export”, the quota ignore mortality events that are potentially numerous during capture and captivity.
- Any capture should be attended and supervised by local authorities (MFMR and/or MECDM) and precisely documented (e.g. timing, location, species, status, sex, measurements, DNA samples, dorsal fin photographs).
- Considering the likely impact on the Guadalcanal population (potentially as much as half of the population was removed), an immediate capture ban is recommended for this population to allow recovery.
- A monitoring program should be developed to document the recovery (or not) of the populations in impacted zones.
- Furthermore, future research effort should extend the study area to cover the entire coastline of Guadalcanal and Malaita.
- A ‘DNA register’ should be developed, i.e. genetic samples of all dolphins captured should be collected systematically and archived to allow verification of its origin and

legitimacy. The Government of SI, the IUCN, on the CITES Secretariat should request samples of Solomon Islands dolphins currently held overseas for genetic analyses.

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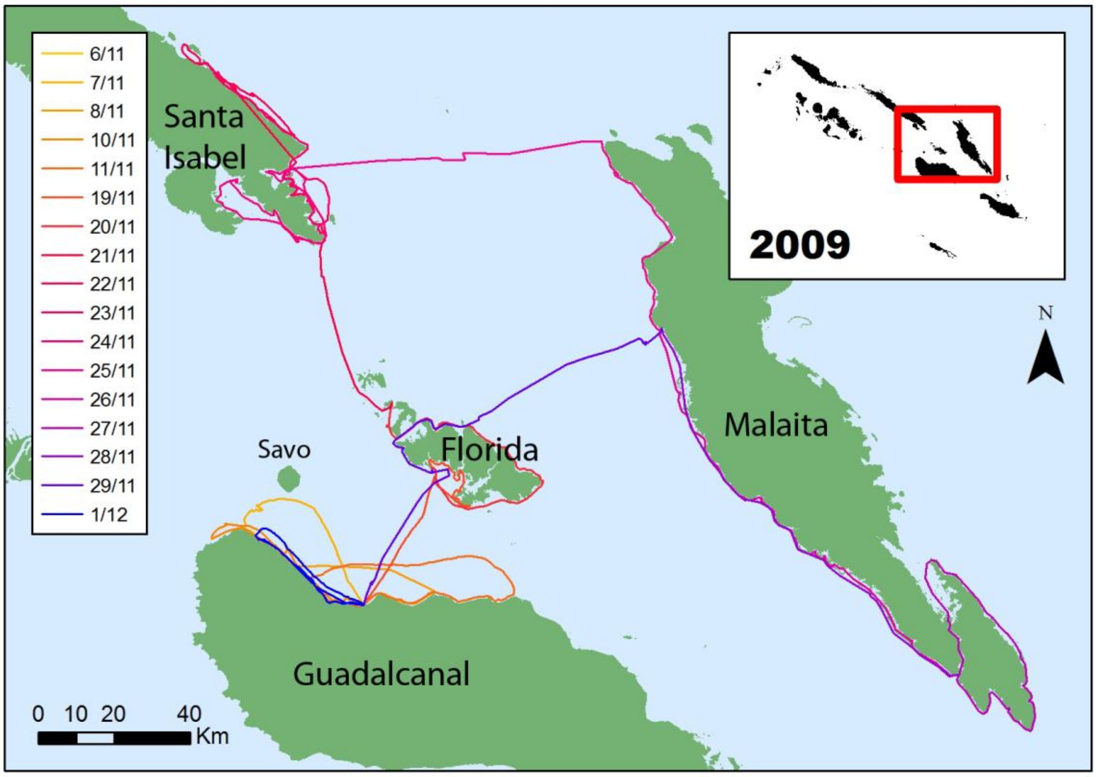
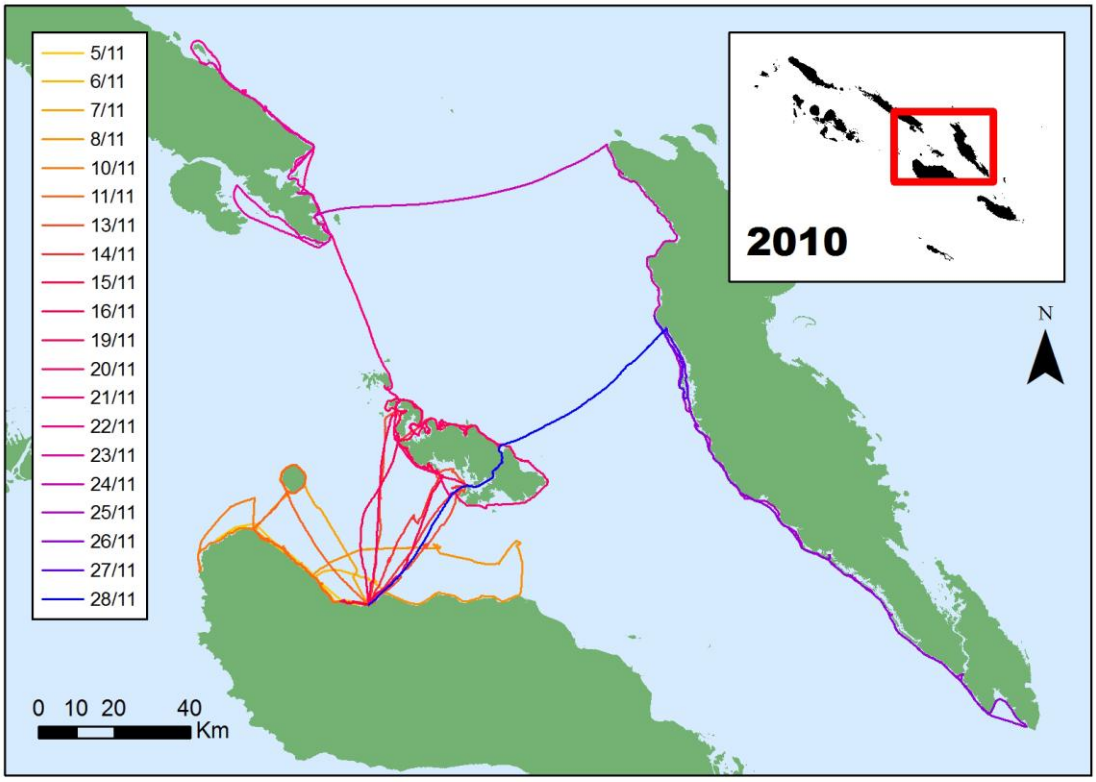
APPENDICES

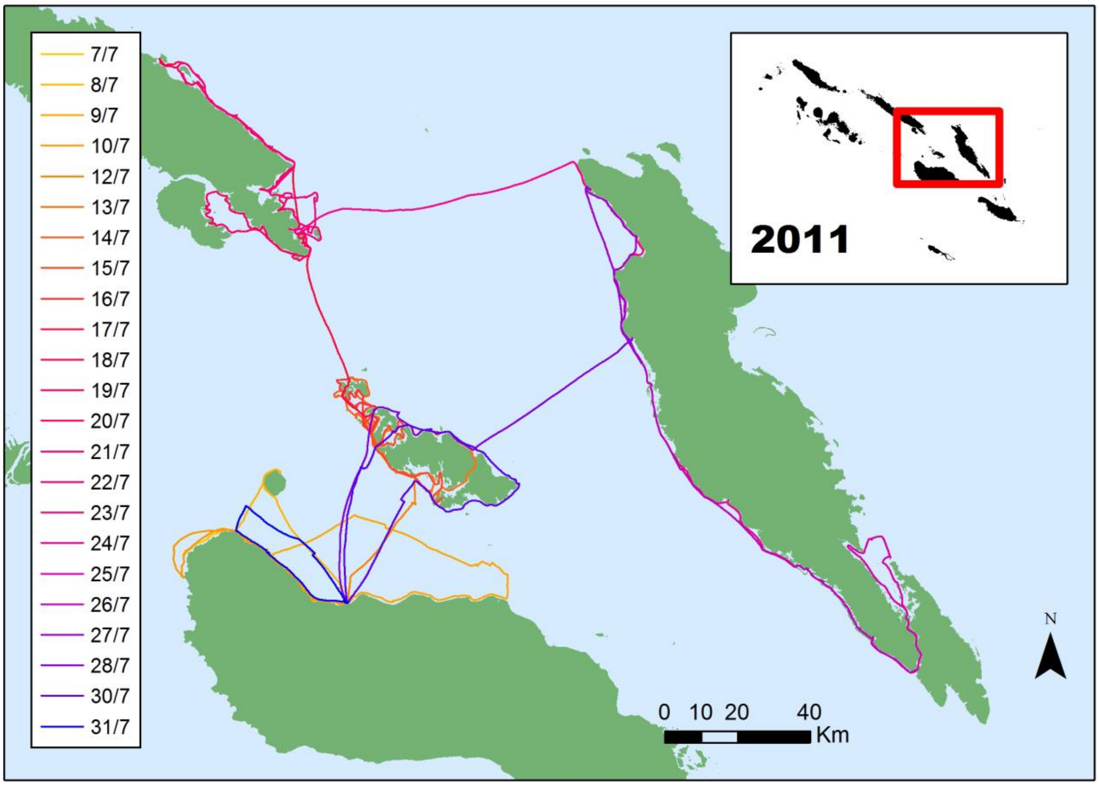
Appendix 1: List of mtDNA haplotypes used in the species identity analyses

Species	Location of origin	# haplotypes	Genbank #	References
Tursiops aduncus	**Solomon Islands**	**7**	**-**	**Present study**
	China/Hong-Kong/Taiwan	13	AF049100/AF056234-AF056243/AF355576-AF355581	Wang et al. (1999), Yang et al. (2005)
	East Australia	6	AF287951-AF287954/EF581128/GQ420670	Moller & Beheregaray (2001), Moller et al. (2007), Wiszniewski et al. (2010)
	Hawaii	1	EF672725	Martien et al. (2011)
	Indonesia	2	AF056237-AF056238	Wang et al. (1999)
	New Caledonia	2	-	Oremus et al. (2009)
	South Africa	4	EF636207-EF636212	Natoli et al. (2008)
Tursiops truncatus	China/Hong-Kong/Taiwan	14	AF056220-AF056232/AF355582-AF355586	Wang et al. (1999), Yang et al. (2005)
	East Australia	3	JN571470-JN571474	Charlton-Robb et al. (2011)
	French Polynesia	2	-	Tezannos-Pinto et al. (2008)
	Hawaii	19	EF672700-EF672718	Martien et al. (2011)
	Japan	21	AB303154-AB303174	Kita et al. (unpub)
	Kiribati	8	-	Tezannos-Pinto et al. (2008)
	New Caledonia	10	-	Oremus & Garrigue (unpub)
	New Zealand	19	EU276389-EU276412	Tezannos-Pinto et al. (2008)
	Palmyra Atoll	7	EF672708-EF672723	Martien et al. (2011)
Tursiops australis	South-East Australia	14	EF192140-EF192149/JN571464-JN571469	Bilgman et al. (2007), Charlton-Robb et al. (2011)

Appendix 2: Boat tracks per year






7/7
8/7
9/7
10/7
12/7
13/7
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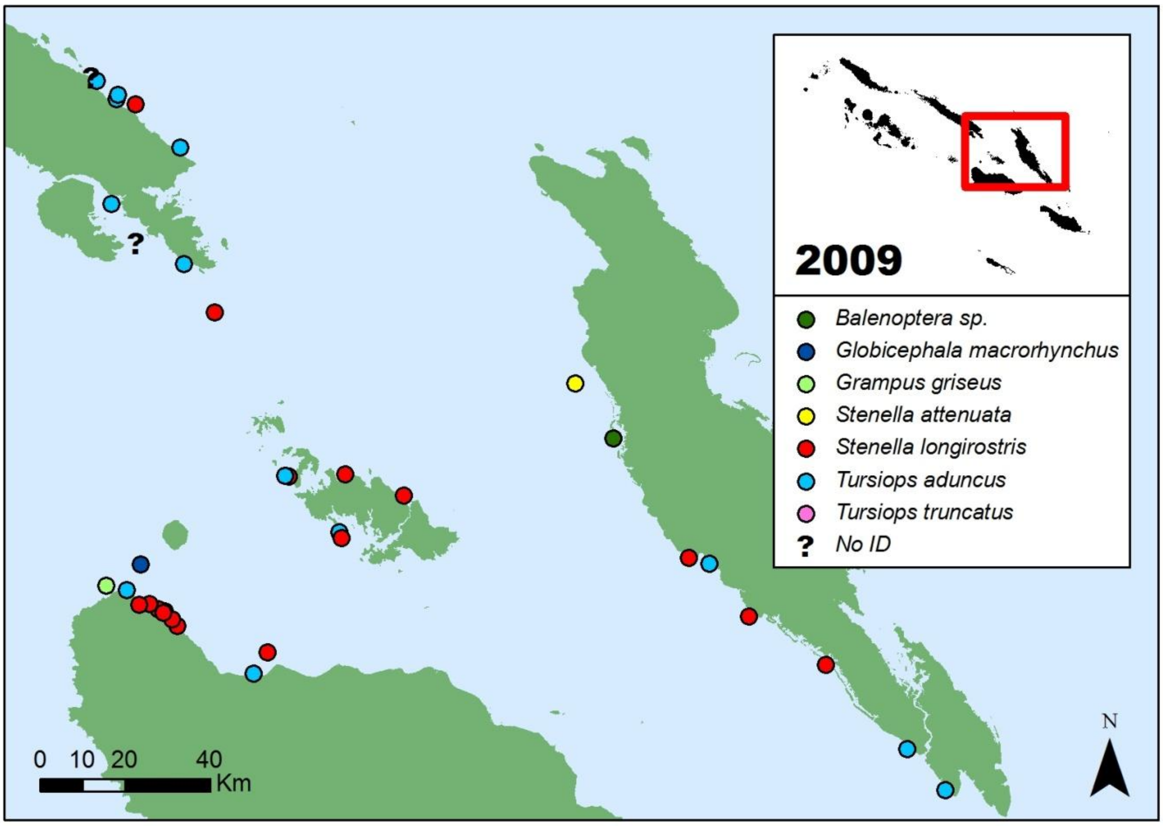
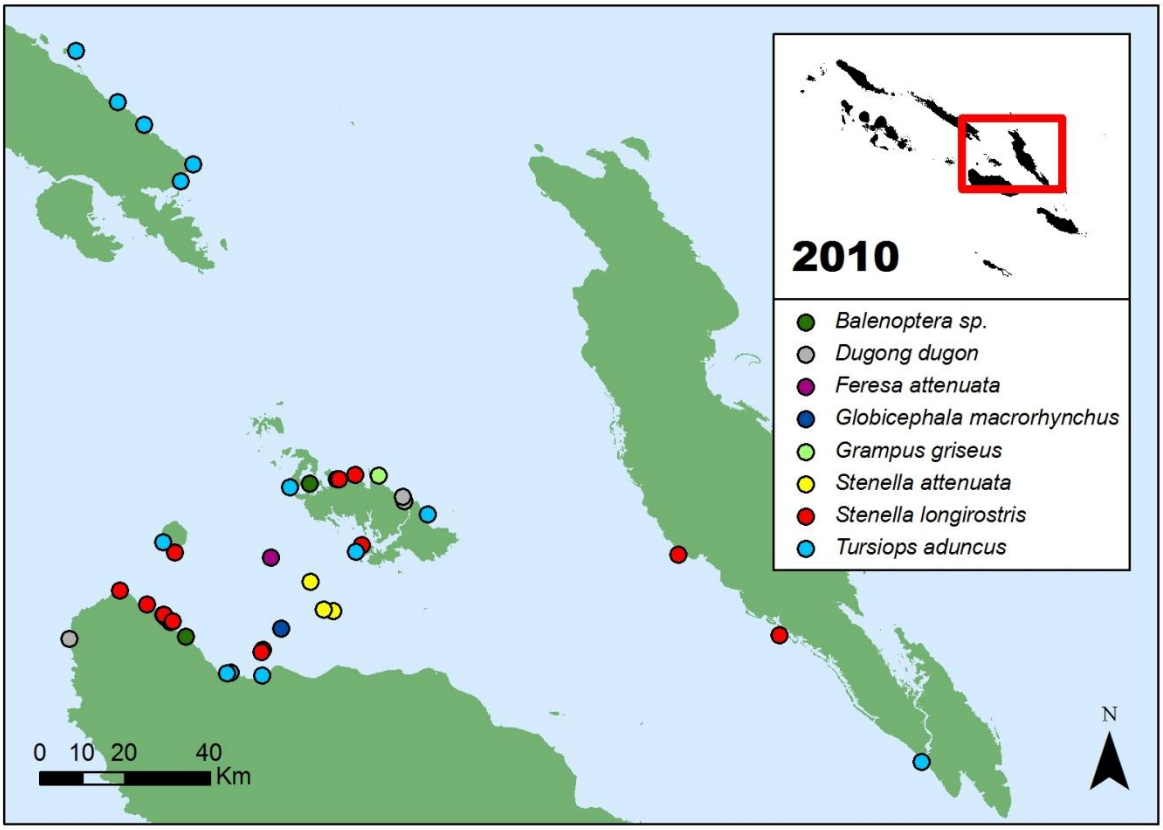
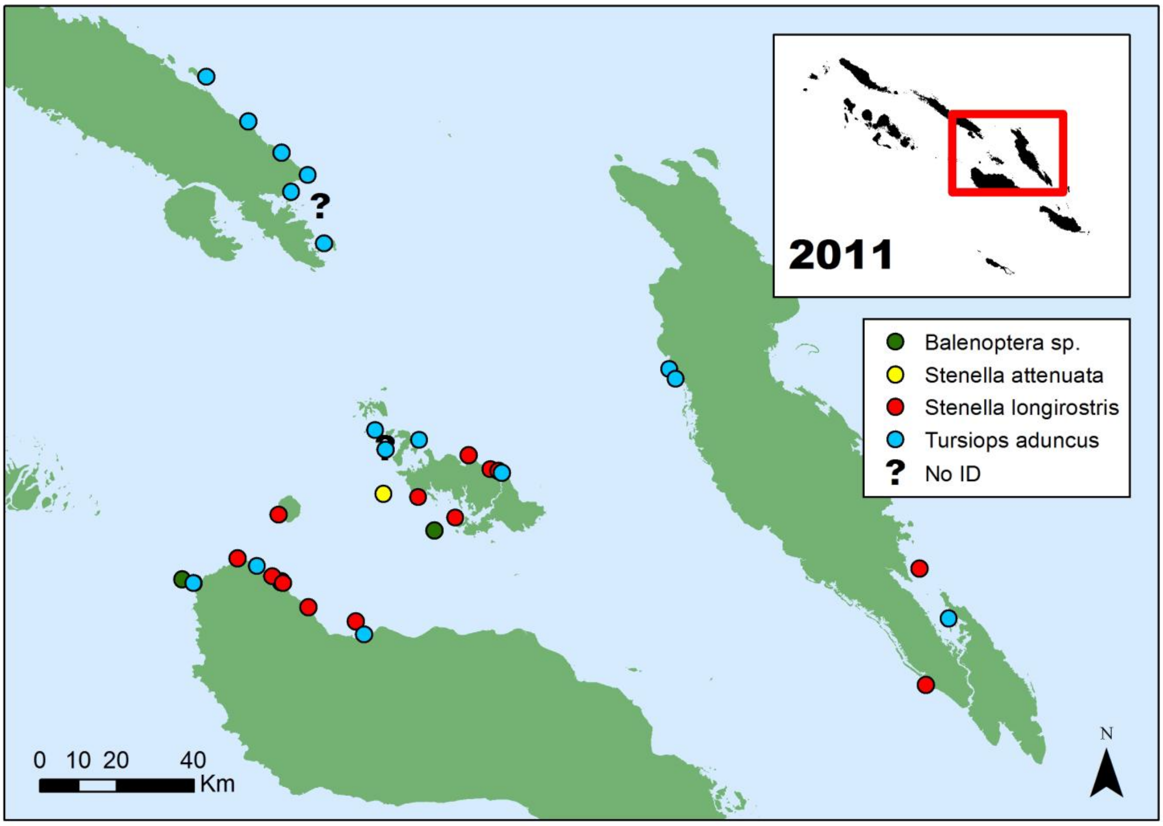
22/7
23/7
24/7
25/7
26/7
27/7
28/7
30/7
31/7
2011
N
0 10 20 40
Km

Appendix 3: Group encounters per year






2011
Balenoptera sp.
Stenella attenuata
Stenella longirostris
Tursiops aduncus
No ID
0 10 20 40
Km
N

Appendix 4: Summary of findings on other marine mammal species

A summary of encounters with the nine species observed during the surveys is presented below. Spinner dolphins (*S. longirostris*), Indo-Pacific bottlenose dolphins (*T. aduncus*) and unidentified baleen whale (*Balenoptera sp.*) were observed at each of the four study sites. They are also the most commonly encountered species, in that order. All three species, as well as dugong, were observed in coastal habitat (Appendix 3). Other cetacean species were typically encountered more offshore (Appendix 3). Although the *Balenoptera* species was not conclusively identified, total estimated length (ranging from 6 to 10m), and photographs of the rostrum and back of the animals suggest that they were Omura's whales (*Balenoptera omurai*). The pantropical spotted dolphin (*Stenella attenuata*) was the next most commonly encountered species and was observed in five occasions. Average group sizes varied among species and were the largest for spotted dolphins, common bottlenose dolphins and spinner dolphins, in that order.

List of marine mammals encountered in Solomon Islands across the study, including number of groups, biopsies and average group size.

Common name	**Latin name**	**# of groups encountered**			**# biopsies**	**Mean group size**
		2009	**2010**	**2011**		
Indo-Pacific bottlenose dolphin	*Tursiops aduncus*	13	16	16	2	10.6 (SD = 10.5)
Spinner dolphin	*Stenella longirostris*	17	18	20	34	57.6 (SD = 58.7)
Baleen whale	*Balenoptera* sp.	1	2	3	0	1.2 (SD = 0.4)
Pantropical spotted dolphin	*Stenella attenuata*	1	3	1	19	87.5 (SD = 58.9)
Short-finned pilot whale	*Globicephala macrorhynchus*	2	1	0	3	36.7 (SD = 17.6)
Dugong	*Dugong dugon*	0	3	0	0	1.0 (SD = 0)
Common bottlenose dolphin	*Tursiops truncatus*	2	0	0	7	60.0 (SD = 14.1)
Risso's dolphin	*Grampus griseus*	1	1	0	0	4.0 (SD = 2.8)
False killer whale	*Pseudorca crassidens*	0	1	0	0	9
Unidentified sp.		0	0	2	-	-

A total of 71 biopsy samples were collected over the three years (32 in 2009, 32 in 2010 and 7 in 2011). Most of these came from two species: *S. longirostris* (n = 39) and *S. attenuata* (n = 20). Results from molecular analyses of these samples will be presented in a separate report.

Information on the status of cetacean populations in the Solomon Islands remains relatively scarce. Some work was conducted in the past by Shimada and Pastene (1995) and Kahn (2006), which provided most of the information available so far on cetacean diversity, distribution and density in the area (Miller 2007). However, nthis study represents the most intensive research effort to date dedicated to gaining information on the marine mammals of the country. All of the nine marine mammal species encountered during our surveys were previously identified in Solomon Islands. Therefore, these are probably common species in this area. It is likely, however, that the description of cetacean diversity in Solomon Islands remains incomplete. Further research effort would be needed to identify additional species, in particular in offshore habitats. The primary focus of this work was to clarify the status of Indo-Pacific bottlenose dolphin population(s) and therefore, the largest part of our effort (73% of distance covered) was made in coastal area where this species is generally distributed. This explains in part why the number of species observed during our study remains limited.

Similarly to the Indo-Pacific bottlenose dolphins, the spinner dolphin was commonly encountered in coastal waters. Spinner dolphins are frequent users of coastal habitats in tropical and sub-tropical areas around the world. However, contrary to *T. aduncus*, they usually do not feed on coastal species since they travel offshore at night to feed on mesopelagic prey. This is to the exception of the dwarf spinner dolphin. *S. l. roseiventris*, the presence of which is not confirmed in Solomon Islands. Further work is clearly needed to clarify the status of spinner dolphins in Solomon Islands since separate populations and potentially sub-species could occur in the area, as suggested by the distinction made by traditional drive-hunt communities in Malaita, which consider three different forms of this species (Takekawa 1996). No clear evidence of these different forms was found during our surveys but ongoing work is looking into this question with the use of molecular tools.

Appendix 5: Recommendations regarding traditional drive-hunt and potential for dolphin-watching tourism

Alhtough the traditional drive-hunt was not the focus of this research, we offer the following recommendations in regards to that issue for consideration by the National Dolphin Technical Committee. The cultural significance of the dolphin drive-hunt in Solomon Islands is widely recognized and the traditional methods have changed little over time. However, there are various reasons to be concerned for the conservation status of dolphin populations in the area. Indeed, the species and numbers taken have been poorly documented but involve sometimes several hundred individuals a year (Takekawa 1996, Kahn 2006). Furthermore, the dynamics of the hunt seem to have varied dramatically through time for unknown reasons (potentially because of temporary dolphin population declines), and key targetted species with the most highly-prized teeth (probably the melon-headed whale) may have disappeared from the region (Dawbin 1966, Takekawa 1996). Therefore, a few initiatives are recommended to improve and facilitate the future management of this practise.

- A research project should be implemented to provide a population assessment of dolphins targeted by the traditional drive hunt. These are primarily spinner dolphins, *Stenella longirostris,* and pantropical spotted dolphins, *Stenella attenuata,* around the island of Malaita.
- Although local knowledge is still limited on spinner dolphin populations in the Solomon Islands, it is recommended that this species should not be hunted unless groups were found in offshore waters (several nautical miles off the coast). Indeed, coastal spinner dolphins tend to form small resident populations throughout the Pacific (Oremus et al. 2007, Andrews et al. 2010) and therefore, care should be taken not to impact such populations that are particularly vulnerable.
- Close collaboration should be initiated as soon as possible between the dolphin hunting communities, the local NGOs and the Government to help document drive-hunt events as accurately as possible. This could be implemented by having observers attending each of the hunting events. Observers would have the role of documenting the species caught, the number of dolphins, morphological measurements and biological sampling (e.g. skin samples for DNA analyses).

- The Solomon Islands Government should encourage and support the development of tourism around wild-dolphin watching as an alternative to live-capture of *Tursiops aduncus*. This study clearly highlights the potential of such activity with numerous areas were the rate of encounters with dolphins is very high.