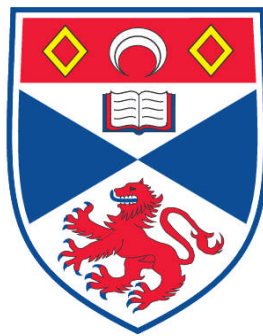# AN INVESTIGATION OF FACTORS RELATED TO THE BYCATCH OF SMALL CETACEANS IN FISHING GEAR

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An investigation of factors related to the bycatch of small cetaceans in fishing gear.

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**A thesis submitted to the University of St Andrews
for the degree of Doctor of Philosophy**

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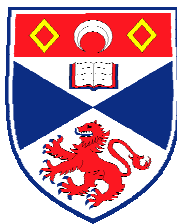
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Abstract

The bycatch of cetaceans in fishing gear is considered to be one of the biggest conservation threats to these species. Gear modifications have the potential to reduce these bycatches in global fisheries but there is little available information on how such modifications may change the fishing performance of gear, or indeed the behavior of cetaceans interacting with fishing gear,

Generalized linear models (GLMs) were used to identify factors related to cetacean bycatches in UK bottom set gillnets. Rigged net height had a significant positive relationship with harbour porpoise (*Phocoena phocoena*) bycatch in ICES Area VII suggesting that lowering the profile of gillnets may have the potential to reduce bycatch rates.

Modifications to gillnets, such as changing the amount of floatation or increasing the density of the meshes, were found to have significant effects on the active fishing heights of these nets. However, results from a bycatch mitigation trial in Argentina showed that the reduced fishing profile of one experimental net did not result in a concurrent reduction in the bycatch rate of Franciscana dolphins (*Pontoporia blainvillei*).

While there was no significant difference in the rate, length or intensity of harbour porpoise encounters in the presence or absence of gillnets, the proportion of fast echolocation click trains were significantly higher when a net was present, indicating that porpoises either increased acoustic inspection of the net or foraging in the vicinity of the net.

An analysis of the underwater video footage collected inside trawl nets in an Australia fishery showed that bottlenose dolphins (*Tursiops sp.*) were present inside nets more frequently than they were caught and were actively foraging inside these nets. The orientation of dolphins inside these nets indicates that the current design of excluder devices used in this fishery could be improved to further reduce bycatch rates.

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1. General Introduction

One billion people currently depend on fish as their main source of protein (Davies et al. 2009). The intense harvest of fish stocks by marine capture fisheries to supply this demand has lead to the decline of many commercial species. This is a trend that is likely to continue, as a growing human population requires more protein from the oceans. In concurrence with an increase in the harvest of targeted catch there has been an increase in the catch of non-target species. These incidental captures are commonly known as bycatch. The United Nations Convention on the Law of the Sea (UNCLoS), UN FAO Code of Conduct for responsible fisheries and the Kyoto Convention encourage nations to implement responsible fisheries management and conservation of resources and aim to specifically reduce discards and catch of non-target species through the development of more selective fishing gears and techniques. In addition, Under the Kyoto Convention, nations are encouraged to increase their efforts in estimating the bycatch and discard rates of species such as marine birds and mammals and to assess the effects of such mortalities on these species (Hall 2000). These bycatches, and their effect on marine ecosystems, can be a major conservation issue (Lewison et al. 2004). In particular, there is growing concern over the effects of such incidental captures on the conservation status of long-lived marine mega fauna such as marine mammals (Read et al. 2006, Read 2008), birds (Zydelis et al. 2009a), turtles (Gilman et al. 2010) and sharks (Baum et al. 2003, Rogan and Mackey 2007). While some species may have the biological capacity to sustain large amounts of bycatch, incidental mortality can be unsustainable for small populations (D'agrosa et al. 2000, Read 2008). The result of such concern has seen an increase in research aimed at monitoring and mitigating bycatch of marine mega fauna since the 1990s (Soykan et al. 2008).

1.1 Marine mammal fisheries interactions

Direct fisheries interactions can have a number of effects on cetacean species. These include enhanced foraging success, changes in distribution and habitat use due to attraction to fishing gear, and an increased likelihood of both entanglement and retaliatory actions from fishermen (Reeves et al. 2001). Interactions are primarily

subdivided into "operational" interactions which occur when there is a direct interaction between an animal and a fishery, (i.e. through depredation or entanglement) and "biological" interactions such as competition between fisheries and marine mammals for resources (Northridge and Hofman 1999).

Bycatch is considered "the greatest immediate and well-documented threat to the survival of cetacean species and populations globally"(Reeves et al. 2005), and it has been estimated that over 300,000 cetacean fishing related mortalities occur globally on an annual basis (Read et al. 2006), with most of these mortalities occurring in gillnets. While the threat of bycatches to cetaceans may be well documented, bycatch rates in the fisheries of many nations remain unstudied and therefore little information exists on the global scale of the problem (Read et al. 2006, Soykan et al. 2008). Bycatch estimates for cetaceans and other taxa, are particularly difficult to obtain for artisanal fisheries (Moore et al. 2010). Significant bycatches have been reported for a number of other marine mammal taxa including the New Zealand sea lion (*Phocarctos hookeri*)(Chilvers 2008), Mediterranean monk seal (*Monachus monachus*) (Karamanlidis et al. 2008) and dugong (*Dugong dugong*) (Read 2008). In addition, while the term bycatch is commonly used to encompass all non-target catch, changes in socio-economic pressures brought about by fisheries collapse, or high bycatch rates of non-target species, have lead to catches that were originally discarded being landed and sold. As an example, this has been reported for Burmeister's porpoises in Peru, which were originally bycaught in a gillnet fishery but became a targeted catch, along with other species such as common dolphins (*Delphinus delphis*) and bottlenose dolphins (*Tursiops truncatus*), and were then used either as bait or for human consumption (Read 2008).

Marine mammal species can also be subject to increased mortality if their direct interaction with fisheries, through the depredation of catch, increases their likelihood of entanglement (Northridge and Hofman 1999, Campbell et al. 2008), or results in retaliatory measures from fishermen (Read 2008). Marine mammal depredation in fisheries has been widely reported (Northridge and Hofman 1999) and is predicted to increase as declines in fish stocks continue (Read 2008). Marine mammals frequently exploit fisheries for food either through direct depredation, foraging on discards or foraging in association with fishing activities. These include depredation from static

nets by bottlenose dolphins (Mussi et al. 1998, Gazo et al. 2001, Read et al. 2003, Northridge et al. 2003) and botos (*Inia geoffrensis*) (Fertl and Leatherwood 1997), depredation or foraging in association with trawls by bottlenose dolphins (Fairfield et al. 1993, Fertl and Leatherwood 1997, Broadhurst 1998, Chilvers and Corkeron 2001, Bearzi 2002, Chilvers et al. 2003) depredation of longlines by a number of cetacean species including pilot whales (*Globicephala* sp.), killer whales (*Orcinus orca*), false killer whales (*Psuedorca crassidens*) and Risso's dolphin (*Grampus griseus*) (Yano and Dahleim 1995, Kock et al. 2006, Garrison 2007, Thode et al. 2007), depredation of traps or pots by bottlenose dolphins (Noke and Odell 2002) and seal species (Lunneryd 1997, Lunneryd et al. 2003, Konigson et al. 2007, Campbell et al. 2008) and associations with aquaculture facilities (Kemper and Gibbs 2001).

1.2   Mitigating marine mammal fisheries interactions

One of the greatest problems faced by managers, fishermen and scientists when trying to find solutions to marine mammal bycatch, is that entanglement events are generally rare, and as a result data relating to such incidents are sparse (Werner et al. 2006). Hall (1996) categorised bycatch events based on one of eight criteria; the spatial pattern; temporal stratification; degree of predictability; whether or not the fishery can control the bycatch event; the type and level of impact; the associated economic and legal implications and the ecological origin of the bycatch. Such categorisation allows specific factors that singularly or collectively may influence the occurrence of bycatch to be highlighted and therefore may also provide insights into best approaches to minimise such events (Hall 1996).

A number of bycatch mitigation strategies have been developed and these can be viewed as following a risk management framework, where the aim of the strategy is to either reduce the risk of exposure of cetaceans to fishing gear or to alter their response to fishing gear when they come in contact with it (Harwood 1999). Examples of such strategies include area restrictions, changes to fishing practices and the use of technical measures. However, for any of these strategies to be undertaken, there needs to be sufficient information on bycatch rates within a given fishery and data relating to both the spatial and temporal occurrence of these events and relationships with

fishing practices or specific gear characteristics. In order to do this, independent onboard observer monitoring programs are required but such programmes are expensive and, if observed bycatch rates are low, many years worth of data may be needed before appropriate management or mitigation strategies can be developed and implemented. While other methods such as strandings data (Iniguez et al. 2003) or rapid bycatch assessments (Moore et al. 2010) can highlight where bycatch problems may exist, most existing bycatch management strategies require data from direct observations of fisheries.

Where a difference exists in the spatial or temporal distribution pattern between the target and non-target species, or there is a good understanding of the spatial and temporal distribution of both non-target species and fishing effort, time area closures can be used to reduce bycatch rates of non-target species. This is particularly the case where oceanographic factors limit the distribution of non-target species (Dunn et al. 2011). For time area closures to work, bycatch events must occur on a relatively small spatial area and be spatially and temporally predictable, and closures should not result in fishing effort being displaced to areas of higher bycatch (Murray et al. 2000). To achieve these criteria adequate pre-closure data on bycatch rates are necessary and for closures to be effective it is important that the industry supports the plan. Unfortunately, in most fisheries, data on the spatial and temporal occurrence of bycatch is sparse or non-existent as is information on the distribution and abundance of the marine mammals they interact with. In addition, for fisheries where there is a high overlap in the distribution of the non-target species and the fishing effort, time area closures may not be economically viable for the fishery to sustain (Hall 2000) and may therefore not be considered as an appropriate management measure to reduce bycatch.

Utilising time area closures to achieve desired goals for single taxa, or species, conservation may in turn have detrimental effects on other species, if these are not taken into account. Baum et al. (2003) showed that an area closure aimed at reducing bycatch of turtles in USA pelagic longline fleets targeting tuna and swordfish in the Northwest Atlantic resulted in a spatial shift in fishing effort and an associated increase in the bycatch rates of a number of shark species. Additionally their results showed that an area closure, which would most effectively reduce the bycatch of

coastal shark species, would result in an increase in the bycatch rates of both turtles and oceanic shark species.

Another method to reduce bycatch is to limit fishing effort, which can be achieved indirectly by placing restrictions on the way in which gear is fished (Read 2000). For example, in New Zealand gillnet regulations to reduce entanglement of Hectors dolphins specify the length and number of nets to be fished, the allowable soak time of these nets, and require that fishermen must stay within 50m of the net while it is fishing (Rowe 2007). Likewise, in the USA mid-Atlantic gillnet fishery, the length of gillnet fleets and number of nets per fleet are regulated as a means of reducing harbour porpoise bycatch (Orphanides 2009).

Acoustic deterrent devices, or "pingers" have also been widely tested as a means of reducing cetacean bycatch. The current generation of pingers were developed in the early 1990s to reduce humpback whale entanglements in cod traps in Newfoundland, USA. The principal behind using an acoustic stimulus was that the noise would either be aversive to cetaceans and therefore keep them away from fishing gear, would encourage echolocation and increase the likelihood of a cetacean detecting the fishing gear, or that cetaceans would learn to associate the sound with nets and therefore perceive this noise as indicating danger (Kastelein et al. 1995b). However, the mechanisms behind how pingers reduce the bycatch rate of small cetaceans are still poorly understood. Currently there are a number of pingers, commercial and otherwise that have been developed and tested to reduce cetacean bycatch, each emitting varying frequencies, pulse intervals and source levels.

The first widespread experiment using pingers in a commercial fishery was conducted in the Gulf of Maine set gillnet mixed fishery in the mid-1990s where a 92% reduction in harbour porpoise bycatch was recorded (Kraus et al. 1997). Since then additional pinger trials in commercial fisheries have also observed a reduction in bycatch rates for a number of marine mammal species, including harbour porpoise (*Phocoena phocoena*) (Trippel et al. 1999, Gearin et al. 2000, SMRU et al. 2001, Larsen et al. 2002), Franciscana dolphins (*Pontoporia blainvillei*) (Bordino et al. 2002), beaked whale species (Carretta et al. 2008), short-beaked common dolphin (*Delphinus delphis*) and California sea lion (*Zalophus californianus*) (Barlow and

Cameron 2003). Pingers have become an integral part in two bycatch reduction strategies in the US for the Gulf of Maine set gillnet fishery and the California drift gillnet fishery (Read 2000) and are mandatory for a number of gillnet fisheries in the European Union (ICES 2009).

Current evidence suggests that, at least in the case of the harbour porpoise, bycatch reduction as a result of pinger deployment is likely to be a consequence of animals being excluded from direct interaction with nets, as a number of studies have shown that harbour porpoise respond aversively to pingers at distances greater than they can perceive nets using echolocation (Reeves et al. 2001). Additionally, harbour porpoise have been shown to reduce their echolocation rate around nets with active pingers (Cox et al. 2001) indicating that the mechanism by which pingers reduce bycatch of (at least) harbour porpoises is not due to them stimulating the animal to echolocate more which should theoretically increase the likelihood of an animal detecting fishing gear.

While the utility of pingers at mitigating bycatch has been proven for a number of species, concerns remain over whether cetaceans will habituate to these devices in time or if their use may result in wide scale habitat exclusion. A number of studies have therefore been conducted to investigate the behavioural responses of cetaceans to pingers. Unlike trials in commercial fisheries, these studies have used different experimental set ups, and have either used simulated nets, or just pingers, or have been investigated on captive animals. An area of exclusion around different makes of pingers has been shown for harbour porpoises (Koschinski 1997, Laake et al. 1998, Gearin et al. 2000, Cox et al. 2001, Carlstrom et al. 2009), Hector's dolphins (Stone et al. 2000), bottlenose dolphins and tucuxi (*Sotalia Fluviatilis*) (Monteiro-Neto et al. 2004). Other studies have focused on whether cetaceans may habituate to pingers, as such habituation may result in an increased probability of entanglement. Habituation of wild harbour porpoise to pingers, defined as a reduction in the exclusion effect of pingers over time, has been shown (Koschinski 1997, Cox et al. 2001). A study on captive harbour porpoise found that displacement to sound playback waned over multiple sessions and in some sessions the animals were observed very close to the sound source (Teilmann et al. 2006). Kastelien et al. (2006) tested the effects of an experimental pinger on a captive striped dolphin and harbour porpoise. While the

harbour porpoise was displaced by the active pinger, an effect which did not wane over the 15 minute test period, no change in distance to the active pinger was noted for the striped dolphin (Kastelein et al. 2006).

While habituation may result in a reduction in the area of exclusion around the pinger the sound stimulus may still "warn" animals of the presence of the fishing net. Therefore, the waning of an avoidance response of harbour porpoise to pingers may not automatically mean that these devices will be less efficient at reducing bycatch. Long term deployment of pingers in the Danish bottom set gillnet fishery has not resulted in an increase in harbour porpoise bycatch (Vinther and Larsen 2004). In the USA, a study investigating the effect of mandatory pinger use on harbour porpoise bycatch rates found that there was no increase in incidental captures over months in areas where pingers were required, and no increase in annual bycatch rates since pinger use became mandatory in the Northeast gillnet fishery (Palka et al. 2008). The authors conclude that these results show no indication that habituation to pingers has occurred in this region. However, they point out that the data analysed did not allow a direct method to investigate habituation, because pingers are not used continuously in any one area and porpoises migrate through management areas and therefore may spend proportions of the year in areas without pingers.

Although pingers are proven to be successful in reducing harbour porpoise bycatch, so far there has been little implementation of these devices in commercial gillnet fisheries outside the USA, where compliance rates have ranged from 3% to 58% (Palka and Orphanides 2008). Factors for slow uptake certainly include the high cost of implementation for fishing fleets and the difficulty in monitoring and enforcement, as well as concerns over pinger durability. In contrast, studies investigating the utility of pingers at mitigating cetacean bycatch in trawls have had mixed results (Northridge et al. 2003, Stephenson and Wells 2006, ICES 2009).

Pingers have also been investigated as a means to reduce bottlenose dolphin depredation of static nets in various regions of the Mediterranean Sea. (Gazo et al. 2001, Brotons et al. 2008, Buscaino et al. 2009). While all these studies reported an increase in target catch and decrease in damage to target catch in nets with pingers deployed, interactions with bottlenose dolphins were not completely stopped by using

pingers. It has been suggested that as bottlenose dolphins use high-intensity, broadband sounds in their intra-specific communication they are less likely to find these types of sound aversive then do species such as the harbour porpoise (Reeves et al. 2001). However, it is likely that the motivational state of an animal will also influence the level of aversion they will have to an acoustic deterrent device.

A study by Cox et al. (2004) investigated the behavioural response of bottlenose dolphins to Dukane NetMark 1000 deployed on a Spanish mackerel gillnet and found that these pingers displaced dolphins in a subtle manner from the net. It was also noted that when the experimental net was hauled dolphins moved towards the vessel and were seen eating discarded fish as well as depredating directly from the net, as evidenced by half-eaten fish. A further trial (Burke 2004) of pingers in this fishery deployed SaveWave pingers that have been specifically designed to reduce dolphin depredation of fishing nets. During the study overall depredation rates were too low to assess whether these devices had any affect on mitigating this behaviour. But, observations of the behaviour and proximity of bottlenose dolphins to the nets was found to be similar in the presence and absence of active pingers. The authors conclude that the SaveWave pingers did not dissuade animals from engaging closely with these nets.

Cox et al. (2004) suggest that the use of pingers may result in an increase in interactions between bottlenose dolphins and nets if exposure to these sounds is positively enforced with an opportunity to depredate directly from nets, or forage on discards when nets are being hauled. There is some evidence that pingers can act as a "dinner bell" and alert marine mammals to the presence of nets and therefore an exploitable source of prey. Bordino et al. (2002) found that during an experiment to test the efficacy of pingers in reducing cetacean bycatch, depredation rates by sea lions were found to be higher in nets with active pingers and this rate increased throughout the length of the experiment.

While time area closures and pingers can reduce fisheries related mortality of non-target species they are only possible under a specific set of circumstances and due to

the high cost related to their implementation and enforcement do not provide an approach that is applicable to small scale artisanal fisheries (Read 2008). Changes in fishing gear and practices are thought to have the highest potential of reducing the bycatch of large marine vertebrates in global fisheries (Werner et al. 2006, Zydelis et al. 2009b). Therefore, the first aim of this thesis is to use statistical models to identify which factors influence the bycatch rates of cetaceans in UK commercial set net fisheries, using independent onboard observer data collected over a 14-year period. In particular, the aim was to see whether any specific gear characteristics were associated with high or low bycatch rates, and whether there is any potential to modify these characteristics to reduce cetacean bycatch in static net fisheries.

1.2.1 Operational changes

A number of fisheries have reduced the bycatch of non-target species by implementing changes in the way in which gear is fished and/or fish discards are managed. For example, the introduction of individual vessel mortality limits in the Eastern Tropical Pacific Tuna fishery led to better skipper performance, and the adoption and improvement of changes in fishing behaviour. In particular, much of the large decrease in dolphin mortality in this fishery has been attributed to the use of the back down procedure during hauling, and the inclusion of a finer mesh "Medina Panel" to further reduce the risk of entanglement (Hall 2000). Individual vessel performance can be improved through selective licensing, for example rewarding "good performers" with licences to fish areas of highest economic values, by enforcing individual catch limits or fleet catch limits, and by marketing labels which identify responsible fisheries to the consumer.

A number of regulatory changes have been made in Hawaiian longline fisheries to reduce seabird bycatch rates. These include changes in the depth, speed and time at which lines are set, changes in bait colour and codes of practice relating to the discarding of offal (Gilman et al. 2008). Changes in the operational behaviour of a fishery to reduce single species bycatch may also result in an increase in the bycatch of other species. For example, in the Eastern Tropical Pacific (ETP) tuna vessels that wish to sell "dolphin safe" tuna set purse seine nets around natural fish aggregating

devices such as logs. Such sets often result in higher levels of elasmobranch and turtle bycatch (Hall 1998). Operational changes must also be those that do not have uneconomical costs associated, are easily monitored or enforced, and are practical and easily implemented by fishermen. Gilman et al (2008) observed smaller reductions in seabird bycatch for two methods (side-setting and dyed bait) than had been observed during experimental trials. The authors suggest that this decrease in bycatch reduction efficiency was likely as a result of changes from experimental methodology that occurred when implemented by the crew under commercial fishing conditions.

1.2.2   Gear modifications

It is estimated that millions of dollars are spent annually on research into the development of fishing techniques and/or gear modifications to reduce or prevent the bycatch of non-target species in commercial fisheries (Werner et al. 2006). To date most of this expenditure has been in North America, Australasia and Europe. Werner at al. (2006) subdivided gear modifications and technical measures into three types. These measures either prevent the non-target species directly interacting with the fishing gear, facilitate escape of the animal if interaction occurs, or allows for post capture release. From a review of the literature Werner et al (2006) estimated that 81% of existing bycatch reduction techniques focus on preventing interactions between wildlife and gear, and of these 61% were specifically developed to exclude non-target species from the fishing ground, gear or bait.

Gear modifications which have been implemented to reduce marine mammal bycatch include the use of Medina Panels in the ETP tuna fishery (Hall 2000), the use of tie downs on gillnets (Murray et al. 2000), restrictions on gillnet mesh size (Rowe 2007, Orphanides 2009) and mechanical alterations to crayfish pots (Campbell et al. 2008). There is also some evidence that changes in the materials used in fishing gear may result in changes in bycatch rates. A study in the US suggests that increase in the occurrence and severity of large whale entanglements with pot fisheries in recent years may be linked to the high breaking strength of modern "polyblend" ropes (Knowlton et al. 2007).

To date, gear modifications and/or technical measures to reduce bycatch have generally developed in response to specific interactions of a single population or species with a specific fishery and generally do not take into account how such measure might effect other species. While the use of circle hooks have been shown to reduce turtle bycatch in longlines (Watson et al. 2005), in some instances they have also been reported to result in an increase in bycatch rates of blue sharks (Werner et al. 2006). A review by Read (2007) showed that while circle hooks have the potential to reduce the bycatch rates of marine turtles in some but not all longline fisheries, and the effects of circle hooks on target catch varied in the four studies he reviewed. The use of tie downs is mandatory for specified gillnets in the US North Atlantic. Tie downs are lines that are shorter than the height of the fishing net and are connected to the float line and lead line of the net at equal distances along the net. Tie downs reduce the profile of the gillnet, and also make the net webbing more "baggy". While their use has been associated with lower bycatch rates of harbour porpoise in gillnets (Palka 2000), bycatch rates of sea turtles (Price and Van Salisbury 2007), and Atlantic Sturgeon (ICES 2011) were found to be lower in nets without tie downs

Gear modifications may also be driven by the economic cost of interactions with marine mammals. For example, the Swedish coastal herring gillnet fishery in the Baltic Sea is thought to be on the point of collapse because of damage caused by grey seals depredating nets and scaring away fish. In this instance knowledge of both the fish and the marine mammal behaviour was used to develop alternative gear (Konigson 2007). Where marine mammals are actively interacting with gear to depredate it, the most successful modifications have physically prevented animals from reaching bait or catch. Campbell et al. (2008) reduced bycatch rates of Australian sea lions (*Neophoca cinerea*) in the Australian west coast rock lobster fishery by fixing a simple t-bar structure inside the pots, which prevented sea lions from depredating catch and from getting trapped in pots. Bottlenose dolphin (*Tursiops truncatus*) depredation in a king mackerel troll fishery in the US was reduced using a prototype low cost simple modification to gear, which dissuaded dolphins from interacting with the catch (Zollett and Read 2006).

1.2.2.1.1 Understanding and reducing cetacean interactions with gillnets.

The majority of cetacean bycatch is believed to occur in gillnets (Read et al. 2006) and one of the postulated explanations for why cetaceans get entangled in gillnets is that they are either unable to detect the nets as they have a low target strength or they detect the nets too late to avoid entanglement. For this reason a number of mitigation strategies have focused on increasing the detectability of static fishing gear to cetacean echolocation clicks by adding passive reflectors, braided rope, air-tube nylon threads, multi-filament threads and increasing the acoustic reflectivity of twine using a chemical filler (Dawson 1991, Dawson 1994, Hatakeyama et al. 1994, Silber 1994, Koschinski 1997). Most recently the focus has been on the development and testing of nylon nets filled with barium sulphate or iron oxide to increase the acoustic reflectivity of the net (Trippel et al. 2000, Trippel et al. 2003, Mooney et al. 2004, Koschinski et al. 2006, Larsen et al. 2007, Mooney et al. 2007). Studies investigating acoustic properties of both barium sulphate and iron oxide net using generated broad band dolphin like clicks and narrowband porpoise clicks found that the target strength (TS) of both nets was greater than comparable nylon nets at or near perpendicular angles, but predicted detection ranges of animals would decrease greatly with an increased angle of incidence to the net (Mooney et al. 2004, Mooney et al. 2007). Mooney (2007) also found that although iron oxide nets had a higher density, they had a lower TS than barium sulphate nets. In comparison a separate study comparing experimental iron oxide cod nets and standard cod nets found no significant difference in target strength between the two materials (Larsen et al. 2007). Although results of some field trials have shown a reduction in harbour porpoise bycatch in chemically enhanced nets (Trippel et al. 2003, Larsen et al. 2007) others have shown no such reduction (Northridge et al. 2003). Furthermore, a study to investigate the echolocation behaviour of harbour porpoise around chemically enhanced gillnets found no difference in the echolocation rates of porpoises around these nets compared to standard commercial gillnets, and concluded that observed reductions in bycatch in these nets was likely to be due to the mechanical properties of these nets rather than their acoustic properties (Cox and Read 2004). However, the effects of such gear modifications on the fishing behaviour of experimental nets have rarely, if ever, been considered in paired experimental trials, and this lack of information makes it difficult

to elucidate by which mechanism bycatch rates have been reduced if gear modifications inadvertently change the way in which experimental nets fish relative to standard nets. In addition, if gear modifications inadvertently change the fishing behaviour of experimental nets the results of such trials may be difficult to interpret, as the probability of entanglement in experimental nets may not be equal to that in control nets. Therefore, the second aim of this thesis is to assess the underwater fishing behaviour of bottom set gillnets with different gear characteristics and to use these data to inform the interpretation of results from bycatch mitigation trials.

In addition to a lack of understanding of how gear modifications may effect the fishing behaviour of bottom set gillnets there remains a paucity of information on the behaviour of cetaceans around these nets and in particular the mechanism(s) by which entanglements occur. It is possible that free swimming cetaceans in the wild may not be so focused in their use of echolocation clicks or may be distracted by prey near or in static gear and may therefore not detect the net. For example, Mooney et al. (2007) found that the acoustic energy reflected from experimental iron oxide and barium sulphate nets was about the same energy as reflected from a 7cm fish, and therefore a larger fish such as a herring would be able to mask reflected echoes from the net. Studies of captive harbour porpoise behaviour around gillnets found that although animals initially avoided the net in their tank, once they had learned to avoid the net they became more easily distracted by prey or con-specifics resulting in an increase in entanglement (Kastelein et al. 1995a).

There is little information available about how odontocete cetaceans use echolocation in the wild and the ecological or behavioural contexts in which it is used (Cox and Read 2004). Bottlenose dolphins have been shown to use echolocation sparingly in the wild and rely mainly on passive listening to detect prey (Gannon et al. 2005). In contrast, free ranging harbour porpoise have been shown to echolocate frequently (Akamatsu et al. 2007) and produce maximum source levels up to 30dB higher than has been recorded for captive animals (Villadsgaard et al. 2007). A diurnal difference in the echolocation rates of free ranging cetaceans has also been reported. For example, some studies have recorded increased echolocation rate of harbour porpoises during the day compared to night whilst other studies have recorded the exact opposite (Cox and Read 2004, Carlstrom 2005). Additionally harbour porpoise have

been observed to forage by "bottom grubbing" where an animal will be positioned with its rostrum close to the seabed focusing its echolocation clicks downwards (Stenback 2006) and it is clear that animals engaged in such a behaviour would have a lower likelihood of detecting a bottom set gillnet before entanglement would occur.

Read et al (2003) investigated the fine scale movements of bottlenose dolphins around commercial Spanish mackerel gillnets in North Carolina, USA using aerial video. The authors defined an "interaction" as when a dolphin was within 500m of the net. The most frequent interaction recorded was avoidance, where animals would change course to move around the net and resume their original course once past it. Such avoidance was recorded at distances up to 100m from the net. Dolphins were also observed begging for fish on a number of occasions and being within one body length of the net though no direct observations of depredation or entanglements were recorded. The authors conclude that bottlenose dolphin frequently interact with these gillnets but rarely become entangled, and when entanglement occurs it is due to the dolphin being either unaware of the net or distracted by other stimuli in the nets vicinity such as fish.

Passive acoustic monitoring (PAM) has been used to investigate echolocation behaviour of bottlenose dolphins around different static net fisheries (Lauriano and Bruno 2007) and harbour porpoise around commercial (SMRU et al. 2001) and modified gillnets (Cox and Read 2004, Koschinski et al. 2006). Results from two of these studies have shown that harbour porpoises are in the vicinity of commercial gillnets more frequently than entanglement occurs (SMRU et al. 2001, Cox and Read 2004). While it has been hypothesised that harbour porpoises may be attracted to struggling fish caught in static gillnets (Gaskin 1984), this hypothesis has not been tested. Therefore, the third aim of this thesis was to use PAM to determine whether echolocation rates of harbour porpoises were higher in the vicinity of gillnets compared to when no net was deployed, and to investigate whether data collected by PAM could be used to determine if harbour porpoise are foraging in the vicinity of nets.

1.2.2.1.2 Understanding and reducing cetacean interactions with trawl fisheries.

While the majority of cetacean bycatches are thought to occur in gillnets, numerous accounts of interactions with trawl fisheries exist. It has been suggested that cetaceans may be motivated to interact with trawl nets as these may present an easy access concentration of prey items which are less energetically costly for foraging cetaceans to exploit (Fertl and Leatherwood 1997). Bottlenose dolphins are the species most often documented feeding in association with trawls (Fertl and Leatherwood 1997, Broadhurst 1998, Chilvers and Corkeron 2001, Chilvers et al. 2003, Scheinin 2010) and reports include observations of direct feeding of fish from codends of shrimp trawls (Broadhurst 1998), and feeding on fish discarded by trawlers (Chilvers et al. 2003). Common dolphins have also been observed to take fish directly from the codend or forage on discarded fish at the surface (Svane 2005). Likewise, pinnipeds (Rowe 2007) and killer whales (Couperus 1994) have been recorded feeding directly off trawl discards. Additionally, killer whales are observed foraging around mackerel nets in the North West Atlantic and the authors suggest whales may obtain a significant portion of their daily dietary requirements by foraging around gear (Luque et al. 2006).

There is evidence that some cetaceans may forage inside trawl nets. Waring et al. (1990) observed pilot whales feeding both around the mouth and inside the net, in the Atlantic mackerel trawl fishery and reported finding mackerel in the stomach contents of bycaught individuals, a prey species not usually found in pilot whale stomach contents. Bottlenose dolphins, pilot whales and Indo-Pacific humpbacked dolphins (*Sousa chinensis*) have been observed feeding around the mouth and inside of trawls (Fertl and Leatherwood 1997) and the remains of target species of the fishery have been found in the stomach contents of bycaught cetaceans

Although interactions between cetaceans and trawl fisheries have been widely documented there have been very few studies on the fine scale interactions between individuals and trawls or the importance of these interactions in the overall foraging ecology of these species. One study, which does look at such interactions in more detail, is reviewed below.

In Moreton Bay, Australia, two distinct communities of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) occur which share an overlapping distribution but are socially segregated and exhibit different foraging behaviours, with one community regularly observed foraging around commercial prawn trawlers, referred to by the authors as "trawler dolphins" whilst individuals of the other community have never been observed to do so (Chilvers and Corkeron 2001). Foraging behaviours recorded in association with trawlers included deep dives in the vicinity of trawler nets or taking discards from trawls. Both communities exhibited different spatial use of the study area, but core areas for each were found to have substantial overlap. Investigation into the behaviour and spatial distribution of these communities in relation to commercial fishing activities found that "trawler dolphins" preferentially forage with trawlers even during the winter season when fishing effort is reduced and nearly 80% of their diurnal behaviour was engaged in foraging (Chilvers et al. 2003). However it is interesting to note that "non trawler dolphins" were also recorded to spend a high proportion of time foraging relative to other studies on inshore bottlenose dolphins, which may indicate a reduced amount of prey availability in the area. Although 64% of "trawler dolphin" sightings occurred when no trawler was present the authors suggest this community is distributed predominantly in waters where trawler operate and noted a seasonal shift in the core spatial distribution closer to the main trawling grounds in summer. In comparison the "non-trawler dolphins" were observed in more shallow, near shore areas where trawling activities do not take place. The authors conclude that the behaviour (foraging or travelling, in relation to season, time of day, tide and presence or absence of trawlers) of the "trawler dolphins" is most heavily influenced by the activities of commercial trawlers whilst the behaviour of the "non-trawler dolphins" was most influenced by season and tide. However, although some trawling occurs in the bay all year round most effort is in summer months, and all trawling is banned at weekends which clearly indicates that "trawler dolphins" do not solely rely on foraging around fishing nets. Trawling started in the region in the 1950s and there are no data available on distribution of bottlenose dolphins prior to this date. Therefore, it is not possible to work out whether the segregation of communities already existed prior to this and "trawler dolphins" adapted to the fishery in their preferential habitat or have changed their habitat preference to exploit the fishery. Chilvers et al. (2003) estimate that one trawler over

10 hours could provide sufficient discards for 5 dolphins energy budget. Kastelein et al. (2003) report that a non-lactating adult female bottlenose dolphin (*Tursiops truncates*) will consume 5kg of fish per day in captivity.

This phenomenon, of only some groups or individuals within populations of cetaceans actively interacting with trawls, has been observed in other studies. Jefferson (Jefferson 2000) noted that only some individuals of pacific humpback dolphins in Hong Kong interacted with trawls whilst others never did, and only some of the resident dolphins studied in the Gulf of Mexico associated with shrimp trawls, whilst others never did (Henderson 2004).

Chilvers and Corkeron (2001) state that entanglement of "trawler dolphins" was extremely infrequent and animals caught tended to be juveniles so were not considered a threat to the viability of the population in terms of management. However exposure to trawling alone may not result in this association as can be seen in areas where not all dolphins associate with trawlers, e.g "non-trawler" dolphins in Australia, bottlenose dolphins in the Gulf of Mexico and indo-pacific humpback dolphins in Hong Kong.

The most commonly used bycatch mitigation strategy to reduce the bycatch of marine mammals in trawl fisheries is the use of excluder devices. Excluder devices are commonly referred to in the literature as SEDs (seal excluder devices), SLEDs (sea lion excluder devices) or MMEDs (marine mammal excluder devices) according to the species interacting with the fishery. The first MMEDs were designed and tested by Gibson and Issaksen (1998) and were loosely based on separator devices deployed in shrimp trawls to reduce large fish and turtle bycatch. They generally consist of a grid placed in an extension in front of the codend, which prevents marine mammals and other large vertebrates from passing into the codend, and instead deflects them towards either a top or bottom opening escape hole in front of the grid. This type of system is used in commercial fisheries to increase gear selectivity and turtle excluder devices (TEDs), which are now compulsory in some fisheries, have been show to greatly reduce turtle bycatch rates in trawls. Sea lion excluder devices (SLEDs) have been used in squid fisheries which operate around the Auckland Islands in New Zealand since 2001 (Rowe 2007). Initially trials found 91% of sea lions reaching the

grid passed though the escape hole out of the net. In the 2004/2005 season of this fishery 8 mortalities were recorded in trawls with SLEDs, of these 7 were a result of animals passing through the SLED into the codend which has resulted in a reduction of the spacing between bars from 28cm to 23cm.

In New Zealand the Hoki Fishery Management Company and the Squid Fishery Management Company have voluntary codes of practice aimed at reducing interactions with pinnipeds which include following best practices during shooting and hauling of nets, stopping discarding during shooting, hauling and when the net is at the surface of the water, and best practice when animals are caught in nets (Rowe 2007).

Mortality of Australian fur seals in the Australia southeast trawl fishery for grenadiers was 34% in nets using seal excluding devices (SEDs) and 78% without SEDs. However, (Hamer and Goldsworthy 2006) in an additional study suggest that as they only observed one seal entering and exiting the net mouth, the observed reduction in bycatch may not necessarily be attributed to the presences of a SED and that most mortalities occurred during shooting of the net. The authors also noted that the increased numbers of seals observed at the surface during hauling is indicative of seals diving to forage on fish in the partially submerged net.

Studies to assess the performance of exclusion grids at mitigating cetacean bycatch have had mixed results (Northridge and Mackay 2005, Stephenson et al. 2006, Lyle and Willcox 2008). While the use of excluder grids has been shown to reduce bycatch rates of bottlenose dolphins in a bottom trawl fishery in Western Australia, the survival rate of dolphins exiting the escape hole is unknown (Stephenson et al. 2006). In addition, while underwater footage of bottlenose dolphins inside trawl nets in this fishery has been collected it has not been analysed in detail to assess whether improvements can be made to the current design and deployment of exclusion grids. Therefore, the fourth aim of this thesis was to investigate by which mechanisms(s) the deployment of excluder devices has reduced bottlenose dolphin bycatch in this fishery and to investigate the behaviour of bottlenose dolphins inside actively fishing trawl nets.

1.3 Summary.

Although there has been a lot of effort in recent years put into the development and mitigation of bycatch of non-target species, especially marine mammals and large marine vertebrates, there is still very little information on the exact mechanism by which bycatch occurs. In the case of cetaceans, pingers have been found to be effective in reducing harbour porpoise bycatch in gillnets but are unlikely to be a feasible mitigation tool on a global level due to high costs associated with purchase, monitoring and enforcement. Currently, work on mitigating cetacean bycatch in static nets is focusing on the acoustic and / or mechanical properties of these nets. However, the mechanisms by which such modifications have reduced bycatch rates remains unclear, (Trippel et al. 1999, Cox and Read 2004, Larsen et al. 2007) and how such modifications effect the fishing behaviour of these nets at sea has not been tested at sea. If gear modifications lead to unintended changes in the probability of bycatch occurring then the results of trials investigating such modifications may be ambiguous.

For fisheries where cetaceans are actively engaged in depredation, acoustic harassment devices and excluder devices have shown mixed results in mitigating bycatch. But it is clear that in fisheries where regular foraging associations exist between cetaceans and trawl nets entanglement occurs at a very low rate relative to the number of interactions. The specific factors of how and why entanglement occur on one occasion and not another are still unknown.

Whilst mitigation measures generally arise out of a conservation, political or economic need to address bycatch issues in specific fisheries, it is clear that in order to develop appropriate long-term mitigation strategies, a greater understanding of how why and when cetaceans interact with fisheries is required. Soykan et al. (2008) divided bycatch mitigation efforts into four distinct categories relating to the behaviour and ecology of the bycaught species, mitigation techniques, social and legal framework available and post-implementation monitoring. Information on the behaviour and motivation of the bycaught species interacting with the fishery is needed. Sufficient bycatch monitoring must occur to identify any spatial, temporal or

gear modifications that may reduce bycatch rates. Any management or technical measures must be developed in line with the fishery and must not incur a prohibitive cost on the economic sustainability of the fishery. Finally, sufficient monitoring must be in place to ensure post-implementation success of any mitigation strategy. However, such steps can only be carried out in areas where legal frameworks and financial support exists, or in high value fisheries where there is impetus to resolve bycatch problems so that the fishery can continue, as is the case in the Eastern Tropical Pacific tuna fishery. The problem becomes more difficult in artisanal fisheries where the value of the fishery is low, but the socio-economic value to fishing communities is high.

1.4 Thesis structure

Chapter 2 of this thesis presents an analysis of 14 years of data on cetacean bycatch in UK static net fisheries collected by the UK onboard marine mammal observer scheme. These data were analysed using statistical models to identify the relationship between different covariates and the bycatch rates of harbour porpoise and dolphin species. The aim of this analysis was to identify gear characteristics which could be tested in experimental trials for their potential at reducing the bycatch of these species.

Chapter 3 of this thesis investigates the underwater fishing behaviour of a number of types of static gillnet during two field trials. While a large number of studies have investigated the species and size selectivity of different gillnets, little information exists on how these nets fish or how small changes in gear characteristics may change this behaviour. Self contained time depth recorders (TDRs) were utilized to measure the headline height of a number of gillnets with different gear characteristics.

Chapter 4 of this thesis used the same methodology to investigate the fishing behaviour of one standard and two modified gillnet types in a gillnet mitigation trial in Argentina. Statistical models were developed to investigate which factors were related to observed bycatch rates using data collected by independent onboard

observers during the trial, and results on the fishing behaviour of standard and modified gear were assessed in relation to results of this analysis.

Whilst chapters 2 – 4 investigated which gear characteristics are related to increased bycatch rates of cetaceans in static net fisheries, and investigated how gear modifications effect the underwater fishing behaviour of these nets, chapters 5 and 6 focused on investigating the behaviour of cetaceans around actively fishing gear.

In chapter 5, passive acoustic monitoring was used to investigate the echolocation behaviour of harbour porpoise in the presence and absence of gillnets, and the effect of float line material on echolocation rates.

In chapter 6, underwater video recordings were used to examine the behaviour of bottlenose dolphins interacting with a bottom trawl fishery in Western Australia. These data were used to investigate the occurrence of dolphins in the vicinity of nets, and the proportion of time spent, and behaviour of dolphins inside trawl nets. These data were then used to investigate the mechanisms by which bycatch in this fishery has been reduced.

Chapter 7 presents a general discussion of the results and conclusions presented in chapters 2 - 6

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2. Chapter 2: Investigating factors influencing small cetacean bycatch in UK bottom set gillnet fisheries.

2.1 Abstract

Generalized linear models (GLMs) with Poisson error distributions were constructed to identify which factors were related to harbour porpoise (*Phocoena phocoena*) and common dolphin (*Delphinus delphis*) bycatch rates in UK bottom set gillnet fisheries. While models combing data collected on hauls in both gillnet and tangle net fisheries were useful to identify fisheries with high bycatch rates of harbour porpoises in ICES Area IV and VII they did not provide information on which gear characteristics might have the potential to be modified as a means of reducing bycatch rates. However, when data for gillnets and tangle nets were modelled separately a number of gear characteristics were found to be significantly correlated to harbour porpoise bycatch rates. Fleet length had a significant negative relationship with harbour porpoise bycatch rates in gillnets and tangle nets in ICES Area IV. Net height and mesh size had a significant positive relationship with harbour porpoise bycatch rates in ICES Area VII. Model results of common dolphin bycatch rates were more ambiguous due to the high spatial and temporal correlation of half of the observed bycatches of this species.

2.2 Introduction

Bycatch in set and drift gill nets remains a principal concern to the conservation status of cetaceans (Reeves et al. 2005), pinnipeds (Goldsworthy & Page 2007), seabirds (Bull 2007), turtles (Gilman et al. 2010) and elasmobranches (Rogan & Mackey 2007). In order to assess the impact of these incidental captures on affected populations, unbiased and precise estimates of bycatch rates are needed (Read et al. 2006). These estimates can best be generated using data collected by independent onboard observer programmes. Such programmes are mandatory in specified static and driftnet fisheries in the USA (Moore et al. 2009) and Europe (Northridge et al. 2007). The total amount of bycatch in a fishery can be estimated using stratified ratio estimation (Belden et al. 2006; Northridge et al. 2007), or through a model based approach (Orphanides 2009; Palka & Rossman 2001; Vinther 1999; Vinther & Larsen 2004).

In order to extrapolate observed bycatch rates to the entire fishing fleet, a measure of fishing effort is needed. Donovan and Bjorge (1995) suggest that the best measure of fishing effort for gillnets is the total length of net in the water multiplied by the total duration of time the nets were soaked. This measure is commonly termed km net hours, and can be used at the individual haul level or grouped at the trip level. These data are rarely available at the fleet level and so other measures of effort are typically used such as number of trips, number of hauls (Northridge et al. 2007) or total weight of fish landed (Orphanides 2009; Palka & Rossman 2001). The large costs incurred with monitoring programmes generally mean that only small proportions of fleets are covered (Northridge et al. 2007; Vinther & Larsen 2004). Some boats may be too small to accommodate observers, and differences in bycatch rates between small boats operating in coastal areas and larger boats operating in open waters may bias bycatch estimates (Vinther 1999). Additionally the same "fishery" may use gear with different characteristics that have different bycatch rates, and fishing practices may change over time.

Data collected by these programmes can also be used to investigate the effects of regulatory measures or mitigation strategies on the bycatch rates of non-target species in monitored fisheries. For example, pingers are required by some gillnet fleets in the US Northeast gillnet fishery. Bycatch rates in observed fisheries in this region were estimated to investigate pinger compliance and the effectiveness of these devices at mitigating harbour porpoise bycatch (Palka et al. 2008). In another study, Gilman et al. (2008) used general additive models (GAMs) to investigate the efficacy of mitigation strategies, such as side setting, in reducing seabird bycatch in Hawaiian long line fisheries. Statistical models can also be used to investigate the spatial and temporal occurrence of bycatch events and the relationship between bycatch rates and gear characteristics or fishing practices. For example GAMs have been used to assess these relationships for harbour porpoise (Palka 2000) and loggerhead turtle bycatch in gillnets (Murray 2009). Palka (2000) used GAMs with a binomial error distribution to investigate which gear characteristics were associated with harbour porpoise bycatch in USA sink gillnet fisheries in the Northeast and Mid-Atlantic. The results of this study showed that the explanatory variables which had the highest correlation with harbour porpoise bycatch were those relating to the spatio-temporal distribution of harbour porpoise, after which the variables net length and net soak duration

contributed most to the fit of the model. Results from this study led to the implementation of regulatory gear restrictions in Mid-Atlantic sink gillnet fisheries. Such studies are important for improving our understanding of how specific gear characteristics or fishing practices relate to bycatch rates. They can also be used to help identify possible mitigation strategies to reduce bycatch, especially those relating to gear modification (Haas et al. 2008). As yet, no such analysis has been conducted on a long-term dataset collected by independent onboard observers in UK static net fisheries.

Both generalized linear models (GLMs) and generalized additive models (GAMs) have been used to generate bycatch estimates, investigate relationships with gear characteristics and to assess the success of mitigation strategies for different bycatch species in different fisheries (Gilman et al. 2008; McCracken 2004). GAMs are extensions of GLMs, but allow the inclusion of smoothing parameters, which allow relationships between the response and covariates to take non-parametric, non-linear shapes. However, while the number of smoothing parameters used can be specified, for small datasets with low numbers of non-zero counts, GAMS can have a tendency to over-fit. Harbour porpoise bycatch in gillnets has been modelled using GAMs and GLMs with quasi-Poisson, "robust" Poisson and binomial error distributions (Orphanides 2009; Vinther 1999; Vinther & Larsen 2004). For modelling bottlenose dolphin bycatch in gillnets a GLM with binomial error distribution has been used (Palka & Rossman 2001).

To produce robust models of bycatch data it is essential that the most appropriate probability distribution is used to model the data. Bycatch can be viewed simplistically to follow a binomial distribution, where bycatch is present or absent in an observed independent unit of fishing effort (trip, haul, set). However, the binomial distribution does not allow the inclusion of count data. Although cetaceans are mostly caught as single individuals per fleet of net, multiple entanglements can occur in the same net (Orphanides 2009). While the Poisson error distribution allows for the inclusion of count data, as bycatch events are generally rare, data collected from observer programmes often result in count data with an excessive number of zero counts. This over dispersion of the data can lead to underestimation of standard errors. To account for this over dispersion, distributions such as the quasi-Poisson, negative

binomial and zero inflated negative binomial (ZINB) or zero inflated Poisson (ZIP) may be more appropriate because no fixed relationship between the mean and variance in bycatch rates is assumed for these distributions. Both negative binomial and ZIP models suit bycatch data where there is a high number of zero counts, and when bycatches do occur counts of individuals are generally very low. ZINB models are useful for data where, along with a high number of zero observations, when bycatches occur, counts can be very large. For example the ZINB distribution was used to model the bycatch of sharks in the eastern Pacific Ocean tuna purse-seine fishery (Minami et al. 2007), where bycatch events containing up to 20 animals were common. Regardless of which distribution is most appropriate, the challenge remains that bycatches are generally rare, and a sufficient number of events are needed in order to fit robust statistical models.

UK static and drift net fisheries target a range of demersal and pelagic species including cod, hake, skate, sole, monkfish and crustaceans, in mesh sizes ranging from 57mm to 356mm and operate throughout the year. Since 1996, the UK Marine Mammal Bycatch Onboard Observer programme, run by the Sea Mammal Research Unit (SMRU), has monitored thousands of trips on commercial fishing vessels around the UK. Trained observers record the incidence of cetacean bycatches in observed hauls as well as information relating to gear characteristics, fishing behaviour and environmental variables. The programme was initially instigated to provide estimates of harbour porpoise bycatch in UK North Sea set net fisheries, but has since expanded to sample a wide number of fisheries and gear types. In 2004, a new EU Council regulation (Reg. 2004/812) came into force, which requires the mandatory monitoring of cetacean bycatch by specified fishing effort for listed fisheries by member states (Council of the European Union 2004). These data have been used to provide yearly estimates of cetacean bycatch in the UK (e.g. Northridge et al 2007). In addition a number of experimental trials investigating different aspects of gear characteristics in relation to bycatch have been conducted (Northridge et al. 2003).

The aim of this chapter is to investigate which factors influence the bycatch rates of cetaceans in UK commercial set net fisheries and in particular, to see whether any specific gear characteristics may be associated with high or low bycatch rates, and

whether there is any potential to modify these characteristics to reduce cetacean bycatch in static net fisheries.

2.3 Materials & Methods

2.3.1 Data Sources

All data were collected by the UK Marine Mammal Bycatch Observer programme, funded by the Department for Environment, Food and Rural Affairs (DEFRA) and the Scottish Government.

2.3.2 Data preparation

2.3.3 Data cleaning and grouping data

A number of hauls recorded in the observer database contained missing values. Where possible, data recorded from hauls within the same trip or from previous trips by the same vessel were used to fill in these missing values. For example, if soak time was not recorded for a haul, the mean soak time of the other hauls recorded in that trip was used. Where there was high variability in soak time between all hauls in the trip, this value was left blank. If fleet length for a haul had not been reported, but the previous trip on that vessel had been observed, then the missing value could be obtained from reviewing the hauls in the previous trip. The same was possible for missing values of mesh size and net height. If either longitude or latitude had missing values, these were approximated from the recorded values of the closest haul. Once as many missing values as possible had been filled in, each haul was assigned to an ICES area, subdivision and rectangle using the package COSTeda in R (R version 2.11.1 © 2010 The R Foundation for Statistical Computing).

2.3.4 Assigning métiers

Observers record nets as belonging to one of six categories: drift net, drift trammel, trammel net, gill net, wreck net and tangle net. However, within these categories there are different net characteristics, such as mesh size and net height, which relate to the commercial fish or crustacean species being targeted. To investigate bycatch rates between different fisheries and to investigate which factors may influence bycatch, these net types were broken down into different métiers.

It is common that an investigation of the most appropriate way to group covariates or split continuous variables into categories is undertaken using classification trees or principal component analysis (PCA). Vinther (1999) used cluster analysis to classify data on target species reported in logbooks to classify trips by métiers. More recently, Fadda (2010) used PCA and cluster analysis to categorize métiers from landings data recorded in the UK marine mammal onboard observer database for 385 trips observed between 2007 and 2009. While Fadda obtained 12 distinct métier clusters using this methodology it was clear that 3 métiers (Pollock – Monkfish, Bass – Ray, Cod – Ray) had arisen from trips where more than one gear type had been fished.

Because data on total fish catches in kg per haul were only recorded for a small proportion of the dataset available to model in this chapter, métiers were primarily assigned to each observed haul using the target catch reported by the observer. This approach was chosen because species-specific gears are typically used to target commercial species in the UK. The range of mesh sizes, soak times and fleet lengths recorded for each of these métiers was then investigated. For those hauls where any of these three gear characteristics was greatly different from the average, further information such as fishing area, year and vessel were used to judge whether a haul should be reclassified as another métier.

Although this methodology uses an *a priori* belief of how different fisheries operate, it is likely that this is a better reflection of how fishing gear is used, than simply using target catch alone as an indicator. For example, when large mesh gill nets used to target sole are damaged or are old, fishermen may use them to target other species such as ray or monkfish. Although they are then used to "tangle" these other species, in a strict sense they are still sole nets and should therefore be characterized as gillnets.

2.3.5   Analysis

2.3.5.1.1   Exploratory analysis of observed fishing hauls

Prior to statistical modelling the variability in the temporal and spatial scale in different drift and set net fisheries was inspected. Data relating to fishing gear

characteristics were assessed for their suitability as explanatory variables of cetacean bycatch, as were the environmental characteristics (depth and sediment type) associated with the observed hauls. Data were also investigated to determine whether any effect of individual observer or vessel on bycatch rates could be assessed. Bycatch rates were calculated for all métiers and GIS was used to produce maps of observer effort and locations of observed bycatch events. This information was then used to aid appropriate splitting of spatial covariates and to assess the outputs of statistical models, so as to ensure that results predicted by models were in the range of those calculated from the raw data.

2.3.5.1.2 Dataset used for statistical modelling

A reduced subset of the entire database was created prior to statistical modelling. In the first instance a number of métiers were removed. Stake nets were excluded from the analysis, because although a harbour porpoise was observed bycaught in this gear, the animal was released alive. Sample sizes for drift nets and drift trammel nets were low and no bycatch was reported in either of these gear types so they were also excluded from analysis. While harbour porpoise bycatch was recorded in trammel nets, the sample size was relatively low and data were missing on the mesh size of the outer net panels used for the majority of hauls of this gear type. Therefore this gear type was also excluded, and the final dataset contained only gillnets and tangle nets. Due to differences in the temporal scale of observer coverage in different geographical areas the data were split by ICES Area (Fig 1) prior to modelling. Finally, to be able to use both forward and backward model step selection, all hauls where values were missing for candidate explanatory variables were also excluded.
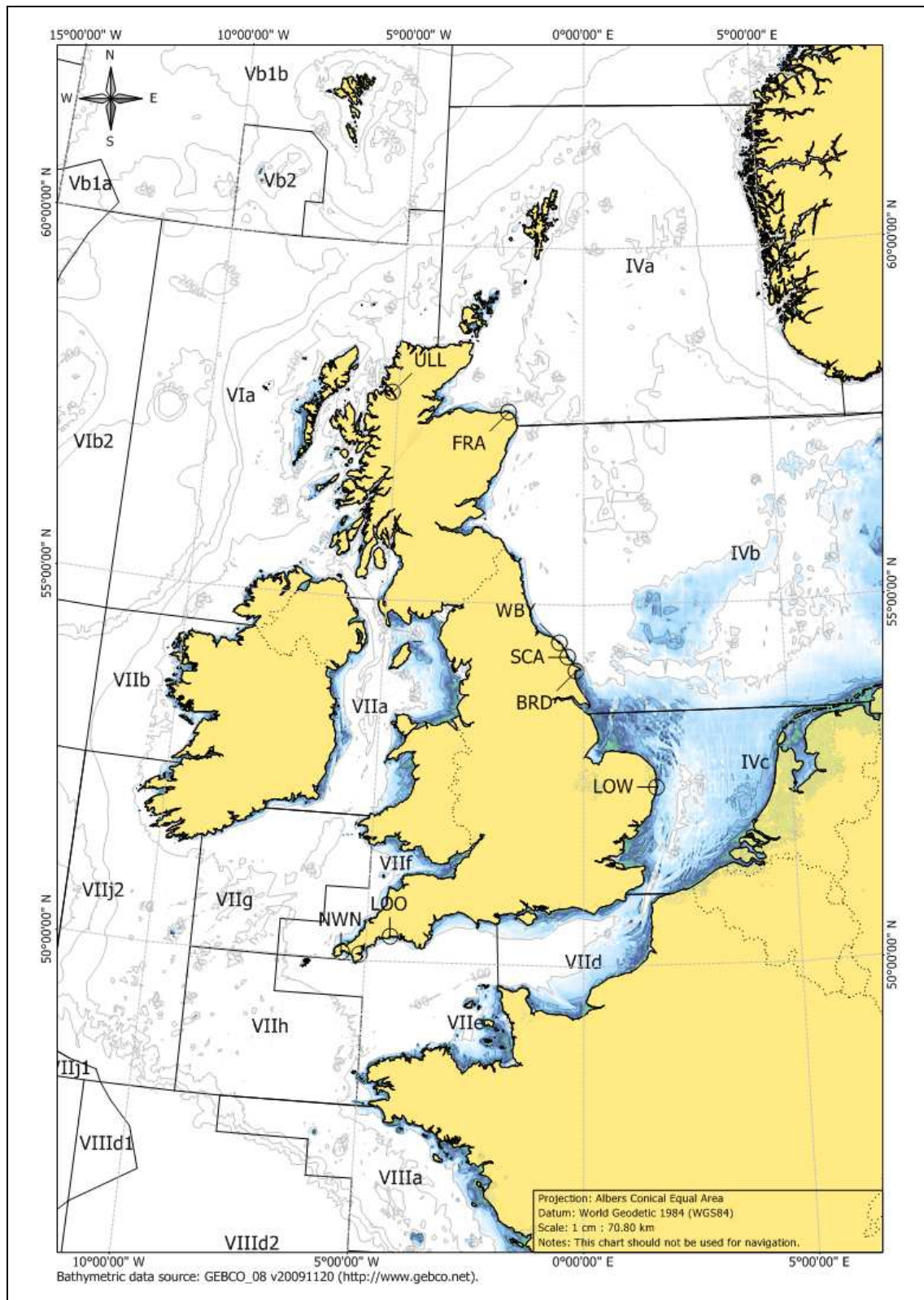
Figure 1: Map of ICES areas and fishing ports. ULL=Ullapool, FRA=Fraserburgh, WBT=Whitby, SCA=Scarborough, BRD=Bridlington, LOW=Lowestoft, LOO=Looe and NWN= Newlyn

2.3.5.1.3 Model structure

The bycatch of harbour porpoise and common dolphins in gillnets can be modelled as the number of animals entangled per unit of effort (e.g. trip, haul, km.net hours). GLMs with a Poisson distribution and logarithmic link function were constructed; such models are commonly used for discrete, non-negative count data (McCullagh & Nelder 1989). The Poisson distribution expresses the probability of a number of events occurring within a fixed time period, under the assumption that observed events are independent and have discrete values with a constant average rate μ.

$$P(\chi) = \frac{e^{-\mu}\mu^{\chi}}{\chi!}$$

Equation 1

Where χ = the number of counts
μ = mean number of successes in a given time interval.

The Poisson distribution assumes a constant relationship between the mean and the variance, with both equal to μ

Fishing effort was calculated as the total length of net deployed times the length of time the net was deployed in the water. The bycatch rate of harbour porpoises could then be modelled using logged fishing effort as an offset in the model:

$$\mu._i = effort_i \; x \; e^{\beta 0 + \beta 1 \chi 1}$$

Equation 2

On the assumption that bycatches are proportional to fishing effort, the offset term (effort$_i$) is included in the model. Because of the log-link this is equivalent to including log effort as a variable in the model with its coefficient fixed to 1.

2.3.5.1.4 Investigation of explanatory variables

Several candidate explanatory variables were selected from the observer database. These related to the spatial and temporal deployment of fishing gear, target catch species and gear characteristics.

As previously stated, fishing effort (km net hours) was included as an offset term in the model, as it seems reasonable to assume that the probability of a cetacean becoming bycaught will be proportional to the amount of net and length of time it is deployed in the water. However, in order to determine whether there are other patterns in the data, which may be related to either net length or soak duration, each of these terms was also offered as a covariate during model selection. Effectively, this means that the effects of each of these variables on bycatch rates can be tested against a null model that assumes proportionality with fishing effort. If either of these terms improve model fit and are significant then it will indicate that the assumption of proportionality does not hold true for the observed fisheries. This is not the same as putting in these terms twice because the parameter associated with the offset has a fixed value of 1 rather than being estimated during the model fitting process.

GAMs (R package mgcv) were run as a way to inspect the relationship between harbour porpoise and dolphin bycatch rates and each explanatory variable. GAMs contain a parameter, gamma, which can be adjusted to reduce the tendency of model over fit. Kim and Gu (cited by Wood 2006) suggested that using a gamma value of approximately 1.4 would reduce this tendency of over fitting without compromising the fit of the model. However, as GAMs were solely used to investigate the relationship between the explanatory and predictor variables, through the visual inspection of GAM plots, gamma was left at the default value 1. Although this means that relationships are less smooth, this allowed the identification of relationships that are truly linear. GAM plots were used to determine which variables had a non-linear relationship with bycatch rates, and whether continuous variables should be split into categorical variables. Covariates that did not show a linear relationship were tested to see if the inclusion of a polynomial term, or adding them as a categorical variable improved model fit of the GLM. The best form of each non-linear covariate was determined by Akaike's Information Criterion (AIC) (Akaike 1973), by constructing simple models with the response term (number of individuals caught), fishing effort as an offset and the covariate being investigated offered as a linear, quadratic and cubic terms. If AIC values were lowest when offered a cubic term the covariate was turned into a categorical variable because problems of collinearity exist in polynomials higher than 2nd order terms (McCracken 2004).

Explanatory variables available for modelling are listed in Table 1.

Variable name
Year
Month
Métier – Defined by the target catch and gear characteristics of each observed haul
Mesh size (mm) – stretched measured mesh size
ICES Area – see Figure 1
ICES Subdivision – see Figure 1
ICES Rectangle – see Figure 1
Latitude – decimal degrees
Longitude – decimal degrees
Soak time - hours
Total fleet length - metres
Vessel length - length overall (LOA), the maximum length of a vessels hull measured at the water line.
Observer ID – categorical by assigned letter code
Depth (m)
Rigged net height – calculated by multiplying stretched mesh size by the number of meshes in height of the net
Presence of floats on float line – yes/no
Experimental haul – yes/no

Table 1: Explanatory variables available for statistical analysis.

Collinearity between variables can lead to unstable parameter estimates and therefore influence the perceived importance of the predictor(s) and lead to poor model selection. Collinearity between variables is likely to exist in data collected from onboard fishery programmes as multiple predictors may be related to the physical properties of specific fishing gear (e.g. mesh size and net height), to fishing practices or there may be temporal and or spatial collinearity because of sampling design. Therefore, prior to modelling, possible collinearity between all explanatory variables was investigated using generalized variance inflation factors (GVIF), conducted with the car package in R. The GVIF threshold for unacceptable collinearity was set at $\geq$ 10. All explanatory variables with a GVIF of $\geq$ 10 were excluded from stepwise model selection.

2.3.5.1.5   Model development and selection

Forward and backward stepwise selection (Venables and Ripley 2002) was used to select the best model for each of the data subsets. This process allows the explanatory power of each of the covariates or interaction terms on the model's fit to the data to be assessed. The process was automated using the step function in R, which uses AIC to evaluate the importance of each covariate to model fit.

The rarity of bycatch events limits the amount of data available to fit complex models, and care therefore needs to be taken not to over-parameterise the model (McCracken 2004). Peduzzi et al. (1996) found that as the number of observed events per variable decreased, the validity of logistic models also decreased. Poisson GLMs with a low mean have similar properties to logistic models, and therefore a cut off point of a minimum of 5 bycaught animals for each covariate retained in the final model was set. During step selection a table of AIC values is produced for each explanatory variable. If the best model had more parameters than this rule allowed, the term with the smallest effect on AIC was removed and step process reran, until the final model did not retain more than the specified allowable number of parameters.

This model structure uses each observed haul as a sampling unit and assumes independence between hauls. However, data collected by the observer programmes are in fact hierarchical, where trips are independent units and hauls within a given trip are related in time and space. If the underlying pattern of bycatch is driven by the relative temporal abundance of animals in an area with fishing gear deployed, then bycatches may occur more clumped than would be expected by chance. If this is the case then bycatch events are not independent, the relationship between the mean and the variance in the Poisson distribution will not be equal, and the data will be over-dispersed. Failure to account for over-dispersion can result in overestimation of precision in the model-based theoretical variances (Burnham & Anderson 2002). Although these data could be aggregated to the trip level to ensure independence, the level of detail required to investigate which gear characteristics or fishing practices are most highly correlated with cetacean bycatch rates could be lost as the type of static net used can vary within a trip. In order to test independence between hauls, the

autocorrelation function `acf` in R was run on the residuals of each of the final models selected.

Over-dispersed count data can be modelled using a quasi-Poisson distribution, where variance is proportional to the mean. The dispersion parameter phi (φ) is estimated from the data. The data are considered to be over-dispersed if phi is >1. However, the level at which phi >1 is truly over-dispersed is dependent on the mean of the data set being analysed. In order to account for possible over-dispersion of the data, each final Poisson model was rerun as a quasi-Poisson. Any model with phi estimated as >1.2 was compared to an equivalent model with a negative binomial error distribution using the function `odTest` in R. This function compares the log-likelihood of a Poisson and negative binomial regression model and can be used to test whether the fixed variance-mean relationship in the Poisson model is true. If the results of this test showed a significant difference between the two models, then stepwise selection was conducted on the data again using a negative binomial error distribution. The best negative binomial model could then be compared to the best Poisson model using AIC.

2.3.5.1.6 Model evaluation

Although residual plots are commonly used to assess the goodness of fit of models, when modelling count data with a small mean and few "distinct values" these plots are less useful. Figure 2 shows the residual plot for a Poisson GLM containing only an intercept that was fitted to a simulated data set generated from a Poisson distribution with a mean of 0.1 (n=1000). This model perfectly describes the process that created the data and therefore the pattern in the residuals is a consequence of the low mean of the data rather than a lack of fit of the model. This pattern, rather than a straight line, is the best pattern that can be achieved from such a dataset. Therefore AIC was used to determine the best model. Once a best model was selected, predicted bycatch rates from this model were compared to observed bycatch rates. If predicted rates were greatly different from observed rates, variables retained in the best model were investigated to identify which covariate(s) was driving the increase or decrease in predicted bycatch rates.
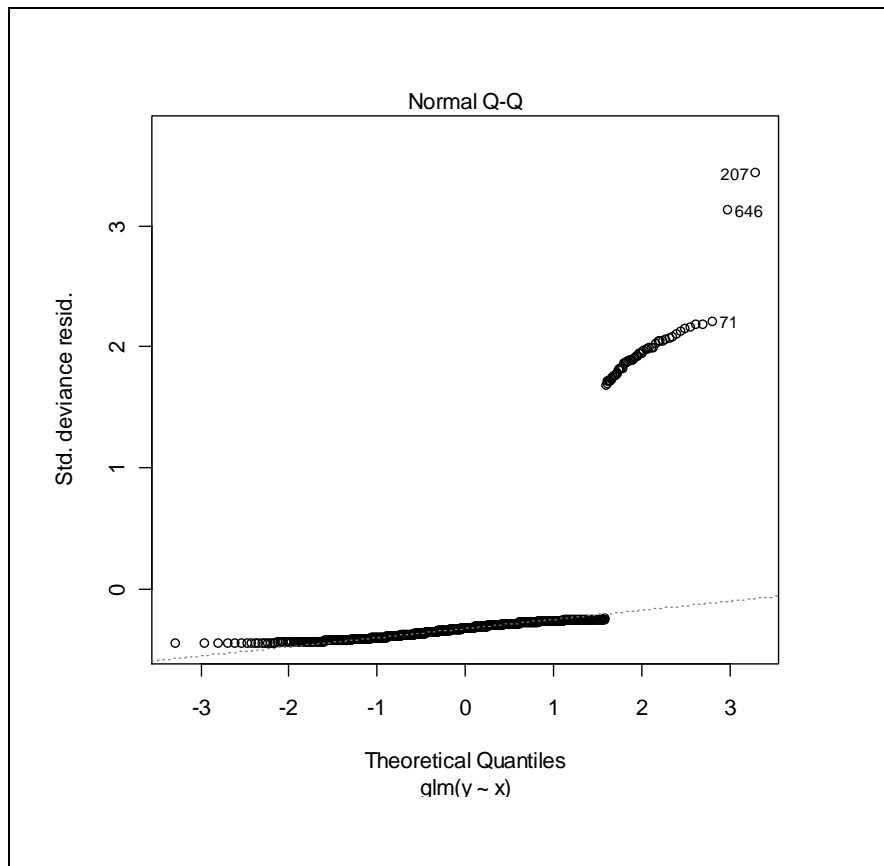
Figure 2: residual plot for simulated data set with mean of 0.1

2.4 Results

2.4.1 Summary of vessels sampled

Between 1996 and 2009, 151 different fishing vessels took onboard observers. Figure 3a shows a histogram of the length in metres for all vessels sampled. The majority of vessels were 10 meters or less in length, as is the case in the fleet overall. The average number of observed trips per vessel was 14, but ranged from a minimum of 1 trip (28 vessels) to a maximum of 171 trips (Fig. 3b). The high number of trips observed on this last vessel was as a result of it being used during experimental trials. Vessels departed from 55 different ports, (Area IV:28, Area VI: 10 and Area VII: 17). The three highest sampled ports in area IV were Bridlington, Whitby and Scarborough and in area VII were Looe, Newlyn and Helford (See Figure 1).
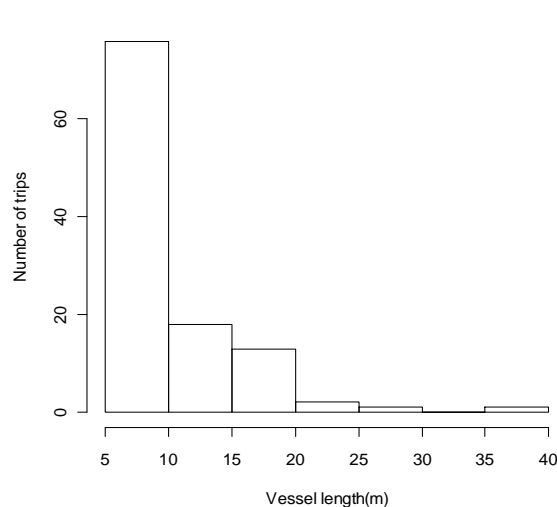
Figure 3a: lengths of observed vessels   **Figure 3b: number of observed trips per   vessel.**
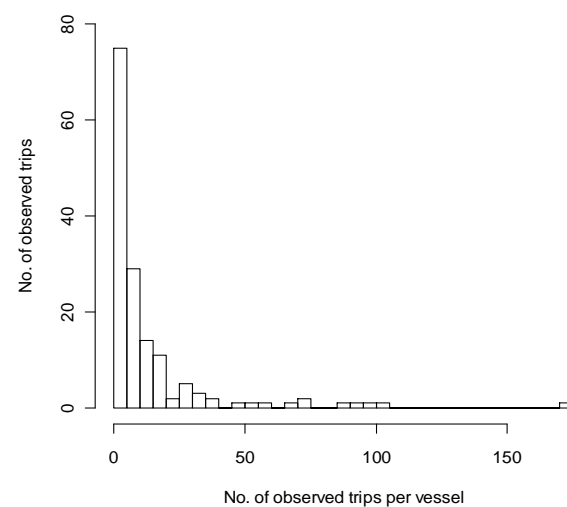
2.4.2 Observer effort

Over the 14-year monitoring period a total of 17 individual observers have been employed and have observed between 24 and 2,415 hauls individually. Figure 4 shows the number of hauls observed by individual observer.
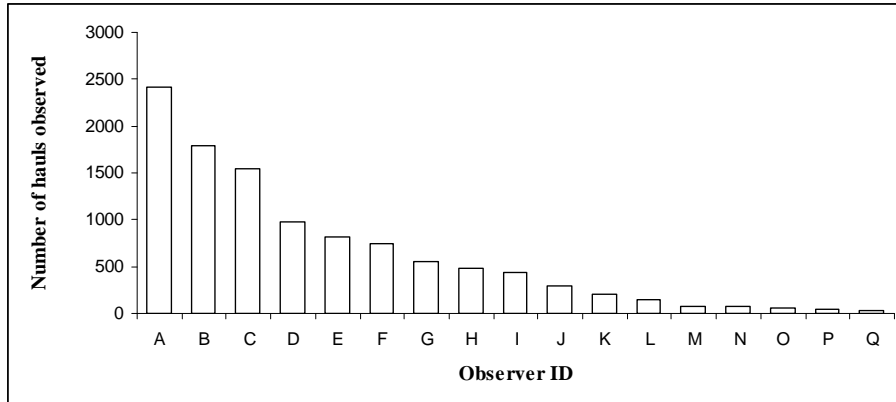
Figure 4: number of hauls observed by individual observers. Observer ID I represents CEFAS subcontracted trips that were conducted by a number of different individuals.

In addition to high variability in the number of hauls monitored by different individuals, observers also worked during different time periods (Fig. 5).
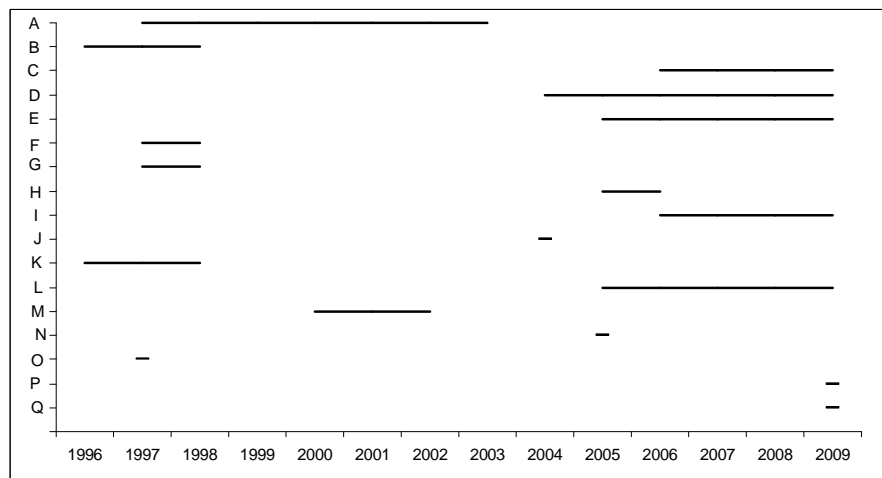
Figure 5: Years monitored by each observer. Yaxis indicates observer ID.

As a result, preliminary analysis showed that observer ID was generally collinear with vessel and year and therefore it was not possible to make a direct investigation of whether any relationship existed between individual observers and recorded bycatch rates.

2.4.3 Summary of observed trips

A total of 1,542 static net fishing trips were monitored between 1996 and 2009 accounting for 2,416 days at sea. During these trips 144 harbour porpoise and 27 dolphins (2 bottlenose dolphins and 25 common dolphins) were observed bycaught. Table 2 provides a summary of the number of trips by the majority of gear types fished. Of these, 214 trips contained 2 net types, 6 contained 3 net types and 1 contained 4 net types. 58% of observed trips were in ICES area IV and 37% of trips were in area VII.

Majority gear type	IV	VI	VII	VIII	Total
Drift net	25	1	20	0	46
Drift Trammel	54	0	8	0	62
Gill net	261	54	229	0	544
Gill net (unspecified)	2	0	6	0	8
Stake net	70	0	0	0	70
Tangle net	220	25	278	1	524
Trammel net	255	0	16	0	271
Wreck net	6	0	11	0	17
Total	**893**	**80**	**568**	**1**	**1542**

Table 2: Summary of the number of observed trips by majority gear type by ICES Area.

2.4.4 Summary of hauls

During the 14-year monitoring period a total of 10,666 hauls have been observed in UK drift and static net fisheries. Figure 6 (a-c) shows the spatial distribution of all observed hauls by ICES Area. Table 3 summarises the number of hauls observed in each ICES area by net type. Unspecified gillnets are those where it was not possible to determine whether these nets were rigged as gillnets or were in fact trammel nets with an unspecified outer mesh size.

Net type	ICES Area				
	IV	VI	VII	VIII	Total
Drift net	79	1	37	0	117
Drift Trammel	227	0	27	0	254
Gill net	2553	330	2251	0	5134
Gill net (unspecified)	23	0	75	0	98
Stake net	168	0	0	0	168
Tangle net	1406	154	1706	20	3286
Trammel net	1352	0	257	0	1609
Total hauls observed	5808	485	4353	20	10666

Table 3: Summary of the number of observed hauls by net type and ICES Area.

54% and 41% of all hauls were observed in ICES area IV and VII respectively.
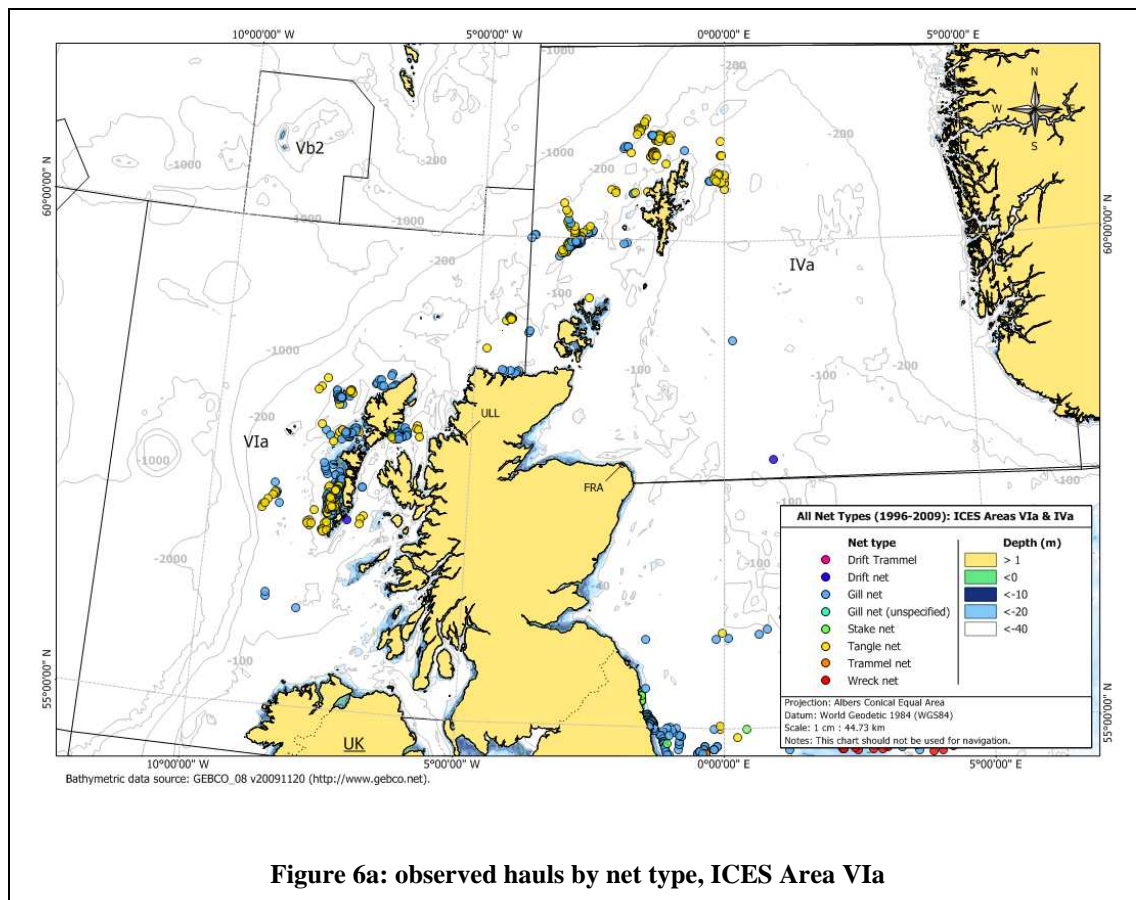
Figure 6a: observed hauls by net type, ICES Area VIa
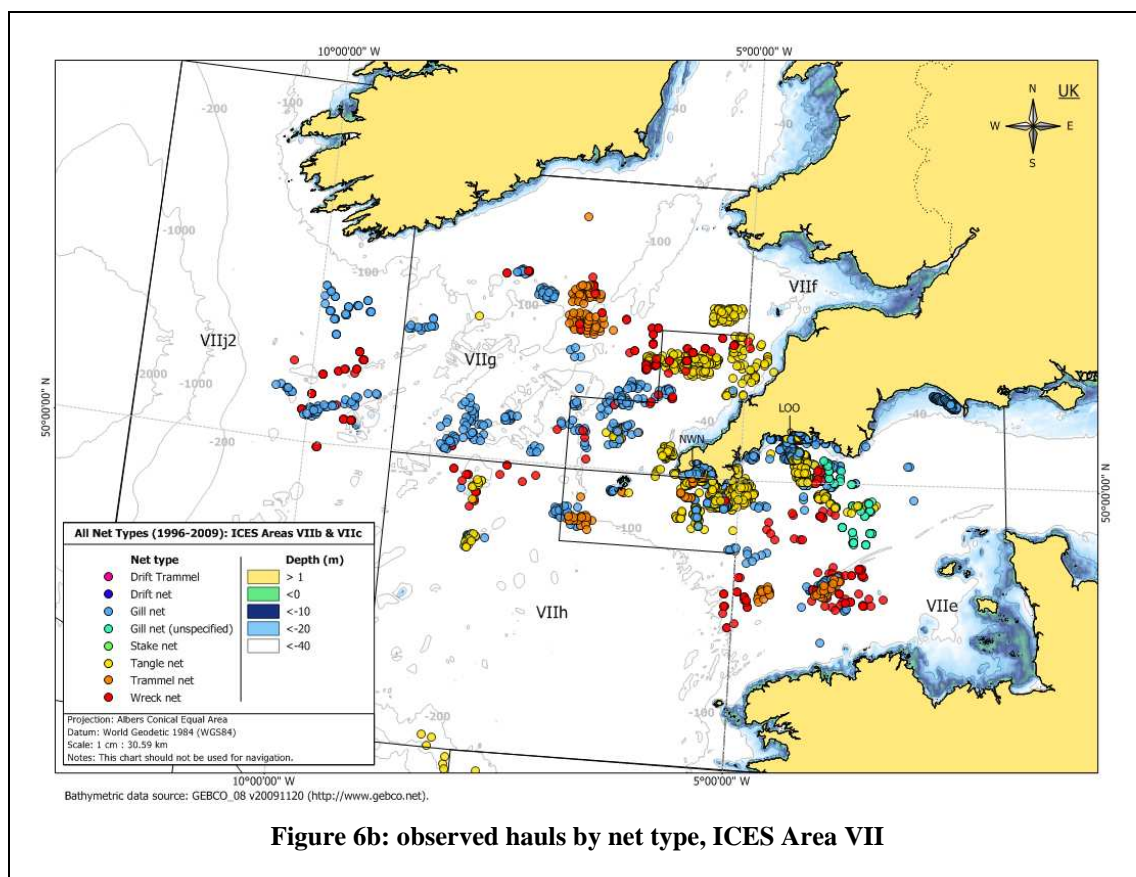
Figure 6b: observed hauls by net type, ICES Area VII
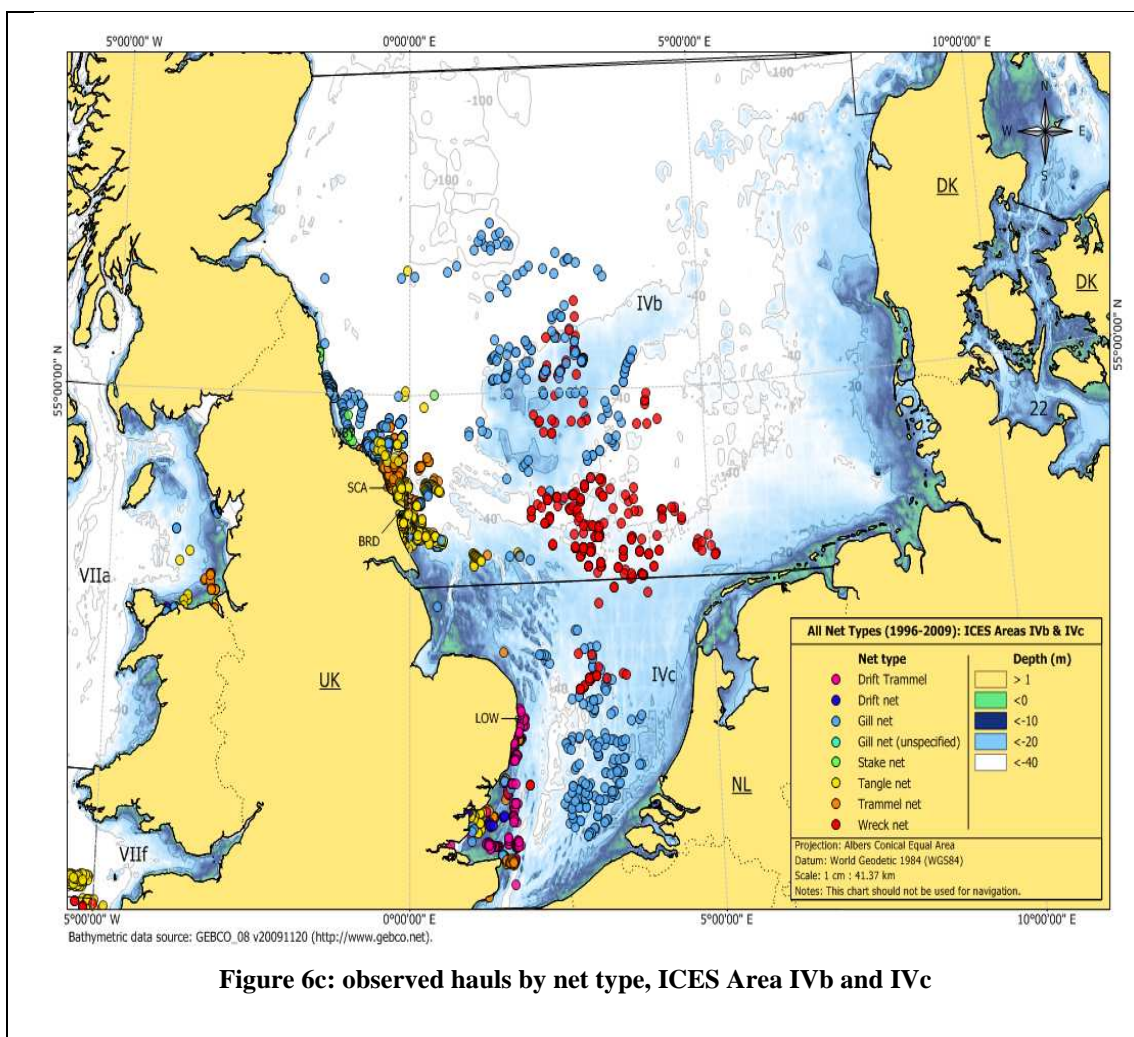
Figure 6c: observed hauls by net type, ICES Area IVb and IVc

2.4.5   Métiers

Using the main target catch *per haul* as the main identifier, a total of 17 métiers were classified for the six net types recorded by observers. Table 4 summarises the target catch species and gear characteristics of each of these métiers.

Métier	Target species	Average mesh size (mm)	S.E.	Average soak time (hrs)	S.E.	Average fleet length (m)	S.E
DN1	Pilchard, herring	64.6	1.6	2.4	0.2	564.8	27.6
DN2	Bass	96.8	0.8	2.6	0.3	542.7	40.3
DN3	Salmon	120.0	0.0	0.9	0.4	550.0	0.0
DRT	Bass, cod, sole, ray	102.9	0.7	1.8	0.4	484.0	12.0
GN1	Bass, haddock	99.5	0.5	19.3	0.5	589.8	20.3
GN2	Cod, ling, Pollock, gadoids	138.8	0.4	17.6	0.3	473.9	10.6
GN3	Hake	122.4	0.3	24.9	0.4	4774.6	80.8
GN4	Dogfish, spurdog	113.9	0.7	20.1	0.5	645.7	11.6
GN5	Mackerel, herring	89.0	19.0	3.0	0.0	900.0	321.5
GN6	Sole, crab, plaice, ray, turbot, monkfish, skate	121.0	0.7	27.3	0.9	763.0	13.2
GN7	Mullet	67.9	0.4	6.3	0.4	334.4	23.5
STK	Salmon, Sea trout	101.8	0.7	5.8	0.4	383.0	7.2
TN1	Brill	212.8	2.1	66.0	14.2	1990.0	268.9
TN2	Ray, monkfish, skate, turbot, dogfish	272.9	0.3	71.2	0.8	1589.0	27.3
TN3	Lobster, crayfish	284.3	2.1	189.6	8.8	741.0	35.9
TR1	Sole, ray, flounder, lobster, turbot, brill, crayfish, monkfish	168.0	3.0	49.1	2.1	1767.0	75.2
TR2	Cod, bass	125.2	0.6	18.7	0.3	402.8	5.5

Table 4: summary of target catch and gear characteristics of assigned métiers.

2.4.5.1 Bycatch by métiers

Bycatch rates of harbour porpoise and dolphins were then calculated, by haul, for each métier. The three highest bycatch rates of harbour porpoise per haul were observed in gillnets targeting hake (GN3), tangle nets targeting species such as monkfish and ray (TN2) and gillnets targeting dogfish (GN4). For dolphin species the highest bycatch rates were observed in tangle nets targeting crustaceans (TN3), gillnets targeting hake (GN3) and tangle nets targeting monkfish and ray (TN2) (Table 5).

Métier	No of hauls	No of porpoise	No of dolphins	Bycatch rate (individuals per haul)	
				Porpoise	Dolphins
DN1	32	0	0	0	0.000
DN2	77	0	0	0	0.000
DN3	8	0	0	0	0.000
DRT	254	0	0	0	0.000
GN1	443	1	0	0.002	0.000
GN2	3215	35	3	0.011	0.001
GN3	348	15	6	0.043	0.017
GN4	345	8	0	0.023	0.000
GN5	2	0	0	0.000	0.000
GN6	657	0	0	0.000	0.000
GN7	124	0	0	0.000	0.000
GNU	98	0	0	0.000	0.000
STK	168	1*	0	0.006	0.000
TN1	8	0	0	0.000	0.000
TN2	3092	74	9	0.024	0.003
TN3	186	3	9	0.016	0.048
TR1	645	5	0	0.008	0.000
TR2	964	2	0	0.002	0.000

Table 5: Summary of bycatch rates per haul per métier for harbour porpoise and dolphin species.*The single harbour porpoise caught in a stake net was released alive and therefore not included in further analysis.

No harbour porpoise or dolphin bycatch was recorded in any of the three drift net métiers or the drift trammel net métier, although effort was relatively low in drift net métiers. Although one harbour porpoise was caught in a stake net in the North Sea, this animal was successfully released alive. Data was missing on the size of the outer meshes used in drift trammel nets for the majority of hauls observed in this metier. Therefore all hauls observed in drift net métiers, drift trammel nets and stake nets were excluded from further analysis. To calculate bycatch rates per fishing effort (km net hour) the data on total fleet length and soak duration of each haul is required. Therefore hauls where either or both of these metrics had not been recorded could not be included in the modelling dataset. In addition, missing values relating to mesh size for unspecified gillnets and the outer walls of trammel nets meant that these hauls were also excluded from the modelling. Table 6 summarises bycatch rates of harbour porpoise and dolphin species, per haul and per km net hour, in all gillnet and tangle net métiers in ICES Area IV,VI and VII.

ICES Area	Métier	No of hauls	Km net hours	No of porpoises	No of dolphins	Bycatch rate per haul		Bycatch rate per 10 km net hours	
						Porpoises	Dolphins	Porpoises	Dolphins
IV	GN1	195	3955	1	0	0.005	0	0.003	0
IV	GN2	1721	10725	19	0	0.01	0	0.02	0
IV	GN4	21	394	0	0	0	0	0	0
IV	GN6	51	784	0	0	0	0	0	0
IV	TN2	1366	53174	53	0	0.04	0	0.01	0
VI	GN2	2	20	0	0	0	0	0	0
VI	GN4	237	2913	5	0	0.02	0	0.02	0
VI	GN6	4	29	0	0	0	0	0	0
VI	TN2	52	2877	2	0	0.04	0	0.007	0
VI	TN3	92	14487	3	0	0.03	0	0.002	0
VII	GN1	215	1572	0	0	0	0	0	0
VII	GN2	926	13187	9	3	0.01	0.003	0.007	0.002
VII	GN3	345	40560	15	6	0.04	0.02	0.004	0.001
VII	GN5	2	3	0	0	0	0	0	0
VII	GN6	508	10473	0	0	0	0	0	0
VII	GN7	114	258	0	0	0	0	0	0
VII	TN1	8	1064	0	0	0	0	0	0
VII	TN2	1438	280372	19	8	0.01	0.006	0.001	0
VII	TN3	89	10260	0	9	0	0.1	0	0.009
Total		7386	447107	126	26				

Table 6: summary bycatch rates per km net hour by métier by ICES Area.

When km net hours are used as a measure of fishing effort, highest harbour porpoise bycatch rates were observed in gillnets targeting cod and other whitefish in ICES Area IV, gillnets targeting dogfish in ICES Area VI and tangle nets targeting ray and monkfish in ICES Area VI. Dolphin bycatch was only observed in ICES Area VII, where highest bycatch rates per fishing effort were recorded in tangle nets targeting crustaceans (TN3), gillnets targeting cod and whitefish, followed by gillnets targeting hake.

2.4.5.2 Experimental hauls

In addition to observer coverage to meet the requirements of the EU Habitats Directive and EU council regulation 812 (Council of the European Union 2004) a number of hauls were observed during a number of experimental trials in ICES Area IV. These trials focused on comparing bycatch rates in static nets with different gear

characteristics, while in ICES Area VII, ongoing trials relating to pinger requirements under EU council regulation 812 are being conducted in tangle net fisheries. In total, 13% of all observed trips in UK static and drift net fisheries (n=212) have been part of these experimental trials. Table 7 summarises the number of observed hauls and observed bycatches during 4 separate experimental trials conducted in ICES Area IV. Results of these trials are reported in Northridge et al. (2003).

Métier	No. of hauls	No. of porpoises	No. of dolphins
GN1	154	1	0
TN2	998	46	0

Table 7: Summary of experimental hauls in ICES Area IV.

Table 8 summarises the number of observed hauls and bycatches during ongoing pinger trials in ICES Area VII. As pingers are known to significantly reduce the bycatch of harbour porpoise in gillnets, all hauls where pingers were deployed (n=26) were removed from the final dataset prior to statistical analysis.

Métier	No. of hauls	No. of porpoises	No. of dolphins	pinger
TN2	69	2	0	No
TN2	26	0	0	Yes

Table 8: Summary of experimental hauls in ICES Area VII.

2.4.5.3 Stratification of data

There was high temporal variability in observer coverage between ICES Areas over the 14 year monitoring period for both gillnets and tangle nets (Figure 7a-b). In addition, although 10 harbour porpoises were observed bycaught in ICES area VI, the number of observed hauls (n=387) meant this subset of data was too small to model. Therefore statistical analysis was restricted to gillnets and tangle net hauls grouped for ICES area IV and VII only.

Figure 7a: observed gillnet trips by ICES area

Figure 7b: observed tangle net trips by ICES area
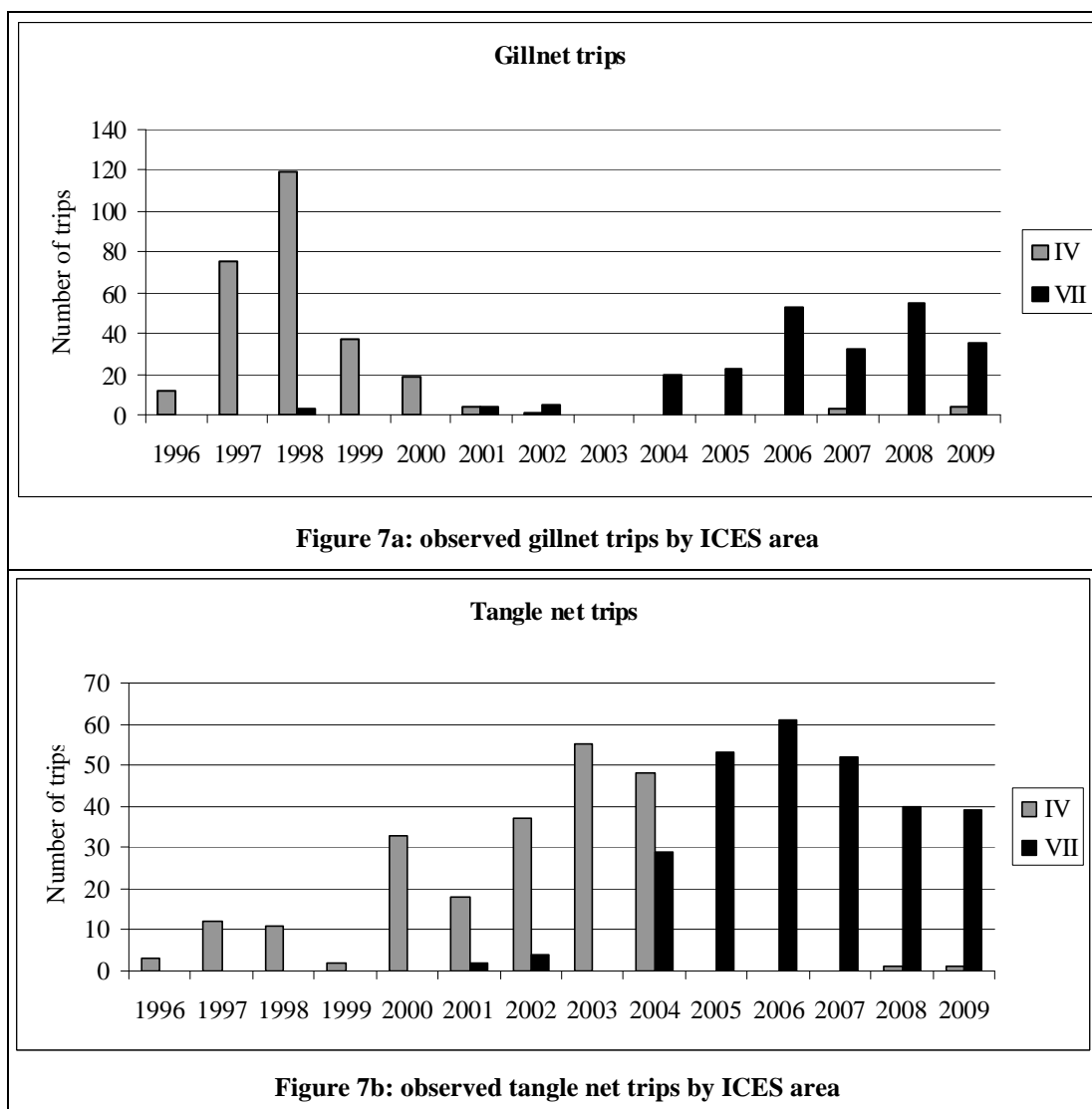
2.4.6 Modelling results

2.4.6.1 Harbour porpoise bycatch in all nets Area IV

After all hauls with missing values were removed, 73 harbour porpoise bycatches in 3,354 hauls observed in ICES Area IV were available for statistical modelling. Table 9 summarizes bycatch rates in the five métiers observed. Highest bycatch rates were in gillnets targeting cod (GN2).

Métier	Observed effort	Number of porpoises	Porpoises per km. net hour
GN1	3955	1	0.000253
GN2	10725	19	0.001772
GN4	394	0	0
GN6	784	0	0
TN2	53174	53	0.000997

Table 9: Summary of harbour porpoise bycatch rates by métier in ICES Area IV

The relationship between explanatory variables and harbour porpoise bycatch rates was investigated visually using GAM plots. Fleet length, soak time, longitude and depth had linear relationships with harbour porpoise bycatch, while mesh size, month and year had non-linear relationships. Simple models of these three latter explanatory variables were constructed, where each model consisted of a single variable. AIC values of this model were then compared when this variable was treated as a linear, quadratic, cubic and as a categorical variable. Both mesh size and month had lower AIC scores when offered as categorical variables, and year had a lower AIC score when offered as a factor. Month was best expressed as quarters (Jan-March, April-June, July-September, October to December) while mesh size was best grouped as meshes < 108mm, 108-149mm, 150-200mm and meshes larger than 200mm. The variable rigged net height had too few unique values to be fitted with a GAM smooth function. However, plots of the raw data indicated that the relationship between net height and harbour porpoise bycatch could be approximated as linear. Observer ID and year were co-aliased terms, meaning they were perfectly correlated. When observer ID was removed, LOA, year, metier and subdivision all had a GVIF >10. The final covariates offered to the model and associated GVIF scores are listed in Table 10.

Covariate	GVIF	Df
log(fleet length)	1.6	1
log(soak time)	2.0	1
Mesh as categorical variable	4.5	3
Floats absent or present	2.6	2
Latitude	2.2	1
Longitude	2.8	1
Depth	2.1	1
Month as categorical variable	1.6	3
log(Net height)	2.2	1

Table 10: GVIF for modelled covariates.

The best model using step wise selection retained the variables fleet length, depth, mesh as a categorical variable, soak time and an interaction between depth and soak time (Table 11). The only significant variable retained by the model was fleet length, which had a negative relationship with harbour porpoise bycatch.

Covariate	Parameter estimate	Confidence interval	P-value
(Intercept)	-1.64	-5.04 to 1.77	p>0.1
log(fleet length)	-0.93	-1.17 to -0.7	p<0.001
Depth	0.04	-0.05 to 0.13	p>0.1
Mesh category (108-449mm)	0.69	-1.50 to 2.89	p>0.5
Mesh category (150-200mm)	1.62	-0.54 to 3.77	p>0.1
Mesh category (.>200mm)	1.62	-0.42 to 3.66	p>0.1
log(soak time)	0.06	-0.58 to 0.70	p>0.5
Depth:log(soak time)	-0.02	-0.04 to 0.004	p>0.1

Table 11: Summary of parameter estimates, confidence intervals and significance levels for all covariates retained by the best model.

2.4.6.2 Harbour porpoise bycatch in all nets Area VII

A total of 3,645 hauls were observed in gillnets and tangle nets in ICES Area VII in which 43 harbour porpoises were reported bycaught. However, bycatch was only recorded in three of the nine métiers observed in Area VII (Table 12).

Métier	Observed effort	Number of porpoises	Porpoises per km. net hour
GN1	1572	0	0
GN2	13187	9	0.00068
GN3	40560	15	0.00037
GN5	3	0	0
GN6	10473	0	0
GN7	258	0	0
TN1	1064	0	0
TN2	280372	19	0.00007
TN3	10260	0	0

Table 12: Summary of harbour porpoise bycatch rates by métier in ICES Area VII

Highest bycatch rates of harbour porpoise (per km net hr) were in gillnets targeting cod and other whitefish (GN2=0.0007), followed by gillnets targeting hake (GN3 = 0.0004) and then tangle nets targeting species such as monkfish and ray (TN2=0.0001). Only 3% of these hauls were observed prior to 2004, during which no harbour porpoise were caught therefore, in order to reduce the effects of any long term temporal relationship with harbour porpoise bycatch rates, modelling was restricted to only those hauls observed since 2003.

Visual inspection of GAM plots showed that the explanatory variables fleet length, soak time and year all had linear relationships with harbour porpoise bycatch rates.

Latitude had a lower AIC when offered as a quadratic term in the simple model. Mesh size was best expressed as a categorical variable (57-113mm, 114-149mm, 150-200mm & >200mm) and month as the categorical variable quarter.

All possible explanatory continuous and categorical variables were inspected for collinearity. Metier was co-aliased with mesh size as a categorical variable. Observer ID, LOA and subdivision had a GVIF value >10. Table 13 provides a summary of the final terms offered to the model and associated GVIF scores.

Covariate	GVIF	Df
log(fleet length)	1.90	1
log(soak time)	4.07	1
Depth	2.45	1
Latitude	1.28	1
Longitude as quadratic term	3.65	2
Mesh size as categorical variable	9.86	3
Month as categorical variable	1.69	3
Year	1.45	1
log(Net height)	3.84	1

Table 13: GVIF for modelled covariates

Using forward and backwards step-wise selection, the best Poisson GLM model to predict bycatch retained the explanatory variables net height, soak time, depth and year. Harbour porpoise bycatch rates had a significant positive relationship with net height and a non-significant positive relationship with year. The relationship with soak time and depth was negative and non significant for both these explanatory variables (Table 14). The model was rerun using a qausi-Poisson distribution, but the estimated dispersion parameter was 1.15 indicating the data were not over-dispersed.

Covariate	Parameter estimate	Confidence interval	P-value
(Intercept)	-430.3	-902 to 42.3	.p>0.05
log(Net Height)	1.785	0.67 to 2.9	p<0.01
log(soak time)	-0.4472	-0.99 to 0.09	p>0.05
Depth	-0.01312	-0.02 to -0.002	p<0.05
Year	0.2102	-0.03 to 0.45	p>0.05

Table 14: Summary of parameter estimates, confidence intervals and significance levels for all covariates retained by the best model.

The results of these models (all nets area IV and all nets area VII) do not provide any insight into which factors may influence harbour porpoise bycatch in this region,

further to the information already provided by estimating bycatch by métier. Therefore the data collected for observed hauls were modelled separately for gillnets and tangle nets hauls for each ICES Area.

2.4.6.3   Harbour porpoise bycatch in Area IV gillnets

Once all missing values had been omitted, the subset of the data for gillnets in ICES area IV comprised 1,998 hauls and 20 bycaught harbour porpoises. Therefore a maximum of four explanatory variables were allowed in the final model. Table 15 summarizes the effort (as number of hauls and km net hrs) for the four métiers observed in this data set.

métier	No. of hauls	km.net hour Effort	No. of porpoises	Porpoise per haul	Porpoise per 10 km.net hour
GN1	195	3955	1	0.005	0.003
GN2	1721	10725	19	0.011	0.018
GN4	21	394	0	0	0
GN6	51	784	0	0	0

Table 15: Summary of harbour porpoise bycatch rates by gillnet métier in ICES Area IV

Investigation of GAM plots showed non-linear relationships between harbour porpoise bycatch rates and soak time, latitude and longitude. A quadratic term improved the fit of these three variables in the simple model based on the AIC score. Both subdivision and longitude had a GVIF>10. Table 16 shows the final variables used in the stepwise selection and their GVIF scores.

Covariate	GVIF	Df
Log(fleet length)	3.45	1
Log (soak time) as quadratic term	3.97	2
Depth	3.78	1
Latitude as quadratic term	3.15	2
Mesh size	3.96	1
Month	1.37	1
Bottom type	2.36	2
Year	1.74	1
Log (Total height)	5.26	1
Presence or absence of floats	2.86	1
Métier	2.90	3

Table 16: GVIF for modelled covariates

Assessing the rug plot for the GAM plot of the explanatory variable latitude showed a lack of observed hauls between latitudes 56 and 59 degrees north. Therefore as well as offering the model this variable as a quadratic term, in order to account for the lack of observed hauls between 56 and 59 degrees north, the model was allowed to determine a slope for each separate area. The best model selected when the variables soak time and latitude were included as quadratic terms had an AIC of 224.1. This model was then compared to the best model chosen by step wise selection when the variables soak time and latitude were offered in the following combinations:

- Soak time as quadratic, latitude with two slopes
- Soak time as linear, latitude with two slopes
- Soak time as linear, latitude as quadratic

The model with the lowest AIC value (220.02) had soak time offered as a linear term and latitude taking two slopes. The best model retained the variables fleet length, latitude, soak time, mesh size and depth, and an interaction between soak time and latitude. Therefore a total of 6 parameters were retained and so step selection was rerun, until the model retained a maximum of 4 parameters. The final model retained the variables fleet length, latitude, mesh size and depth (Table 17). The last variable removed from the model prior to this final step was soak time. The difference in AIC values between the best model and the model including soak time was 0.36 indicating very little improvement over the 5-parameter model.

Covariate	Parameter estimate	95% Confidence interval	P-value
(Intercept)	-22.11	-34.08 to -9.07	p<0.001
Fleet length	-1.65	-2.47 to 0.79	p<0.001
Latitude	0.42	0.15 to 0.68	p<0.01
Depth	-0.04	-0.08 to -0.01	p<0.5
Mesh size	0.03	0.00 to 0.05	p<0.5

Table 17: summary of parameter estimates, confidence intervals and p-values for the best model.

The model was rerun using a quasi-Poisson error distribution. The dispersion parameter φ was estimated at 1.01, indicating that the data were not over-dispersed. Figure 8 shows predictions of harbour porpoise bycatch rates at interim values of fleet length (549m), mesh size (120mm), latitude (55 degrees north) and depth (32m).
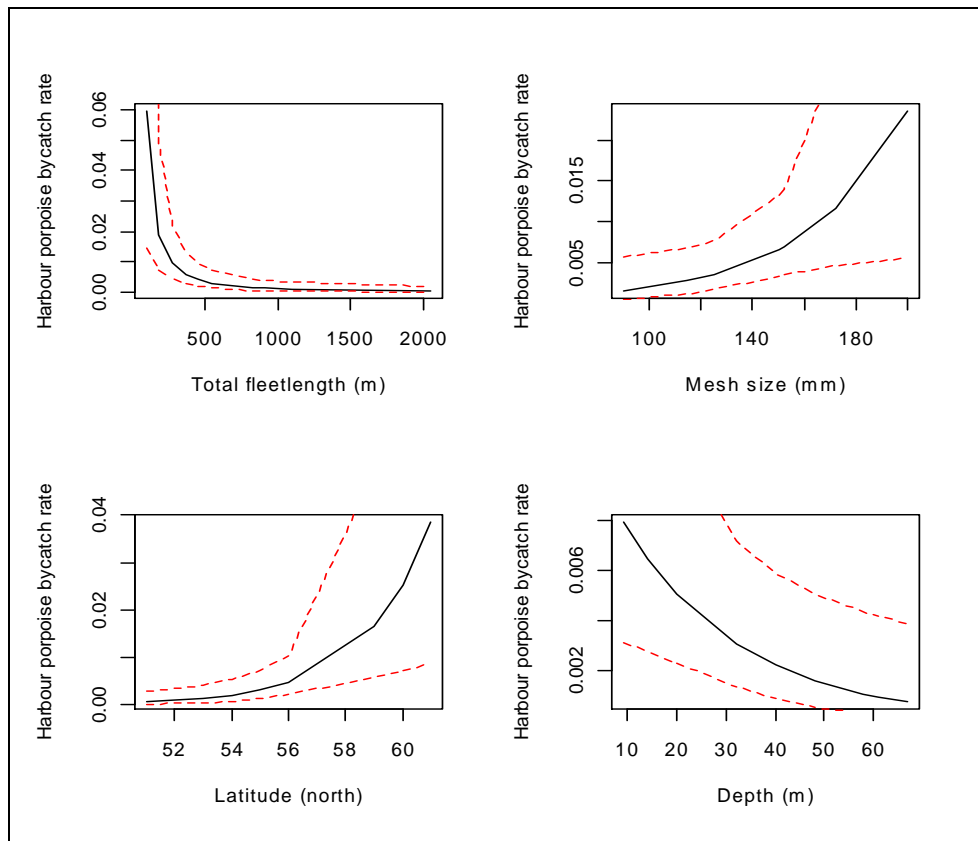
Figure 8: Plots showing the relationship between predictor variables and harbour porpoise bycatch (offset by km.net hour). The red dashed lines indicate the 95% confidence intervals. N.B .y-axis scales are different.

The negative significant relationship between harbour porpoise bycatch rates and fleet length as predicted by the model are driven by the high bycatch rate (per unit effort) in fleets of length 550m or less (Table 18.).

Fleet length	Métier	Number of hauls	Effort (Km. net hours)	No. of harbour porpoise	Bycatch rate by haul	Bycatch rate by effort
<550m	GN2	1570	8124	16	0.010	0.002
>550m	GN2	151	2601	3	0.020	0.001

Table 18:Summary of harbour porpoise bycatch rates in short and long cod nets.

These shorter nets are known as wreck nets, and as their name suggests, are typically shot over wrecks or over rough ground. Observed wreck nets had a larger mean mesh size than longer nets targeting cods (Fig. 9). No harbour porpoise were caught in depths greater than 65m.
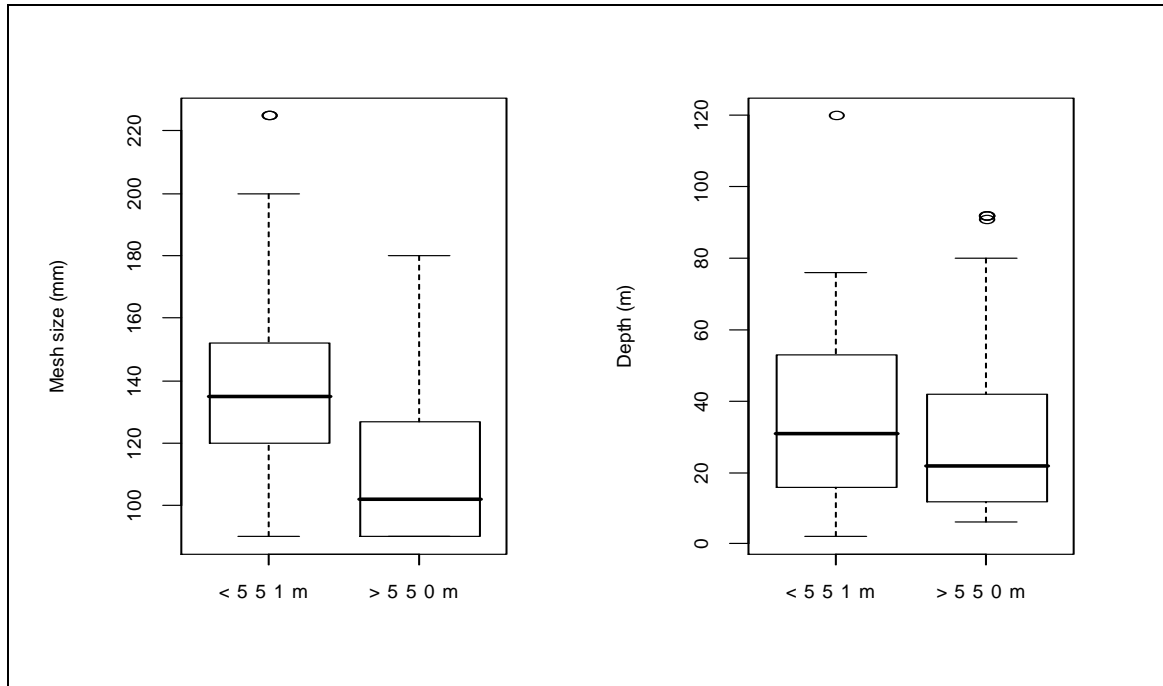
Figure 9: Box plots of mesh size and depth fished for observed hauls in cod wreck nets (fleet length <551m) and longer cod nets in ICES Area IV.

Figure 10 shows the locations of all observed hauls that targeted cod in ICES Area IV, colour-coded by fleet length ($\geq$ or $<$ 550m). The significant positive relationship between harbour porpoise bycatch rates and latitude is driven by higher bycatch rates as latitude increases, in wreck nets in the central North Sea and in cod nets longer than 550m observed west of the Shetland Isles.
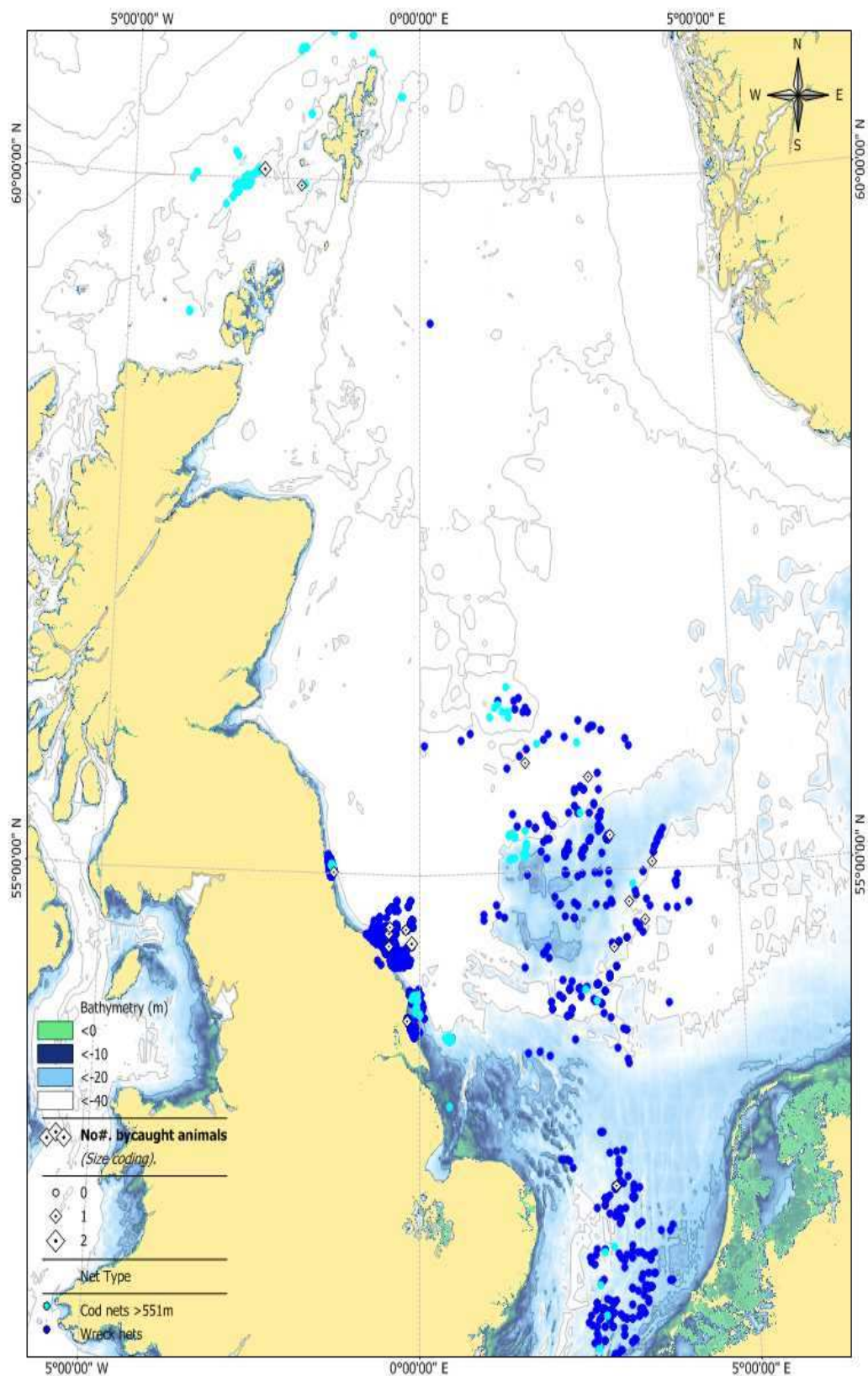
Figure 10: location of observed gillnet hauls targeting cod in ICES Area IV. Darker blue circles indicated observed hauls in cod wreck nets.

Correlation of bycatch between hauls was investigated using the acf function in R. The residuals of the final model were permuted 100 times using this function to find the 95% confidence intervals for the data set. Although the function acf produces a graph with a dotted line representing the 95% confidence intervals, this is based on an assumption that the data follow a normal distribution that is unlikely to be upheld by a Poisson distributed dataset with a low mean. Investigation of residuals from the final model showed autocorrelation at a lag of one indicating bycatches in subsequent hauls occurred more frequently than by chance (Fig. 11)
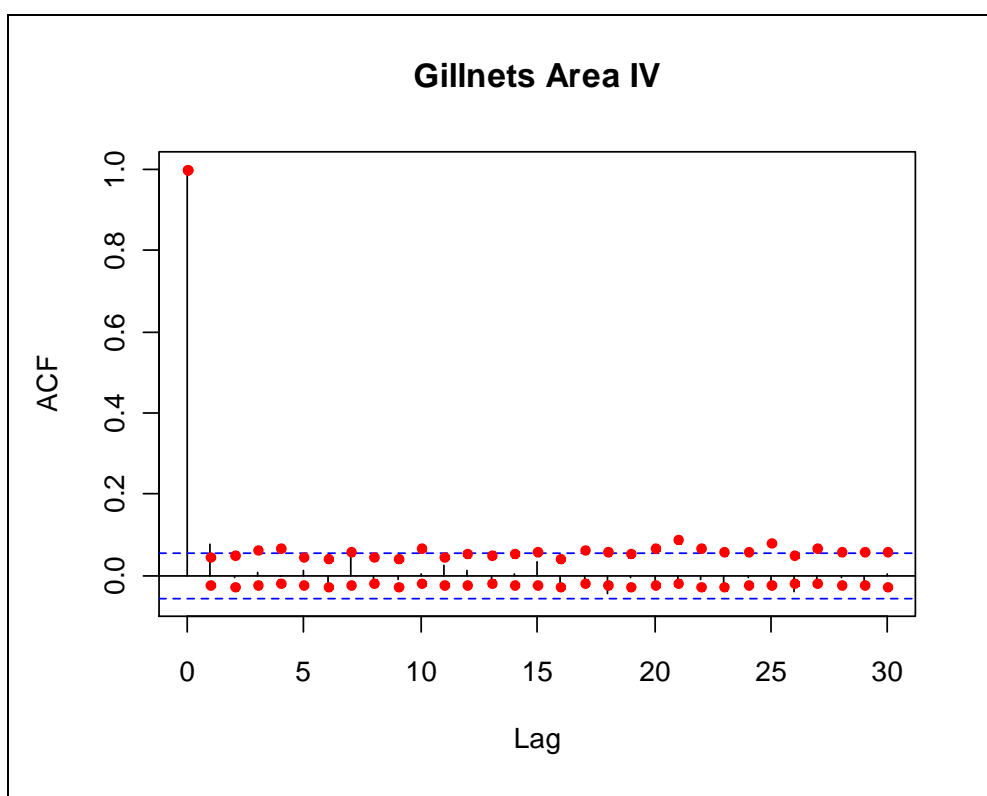
Fig 11: autocorrelation of bycatch events in subsequent hauls within trips. Red dots indicate 95% confidence intervals generated after re-sampling the final model 100 times. Horizontal lines show the correlation at each lag.

Of the 20 observed bycaught harbour porpoises, one was caught in a bass gill net (métier GN1) during an experimental trial. All other porpoises were observed bycaught in gillnets targeting cod (GN2). A single porpoise was caught in a single haul in eight trips, two porpoises were caught in a single haul in one trip, two porpoises were caught in two separate hauls in two trips and one trip caught three individual animals in three separate hauls.

2.4.6.4   Harbour porpoise bycatch in Area IV tangle nets

In Area IV, 98.7% of hauls in tangle net fisheries were observed prior to 2005. No hauls were then observed until 2008. To remove any temporal effect on bycatch rates data were therefore restricted to hauls observed prior to 2005. A total of 1,356 hauls were available to model, with an observed bycatch of 53 harbour porpoise (Table 19). Therefore 10 parameters were the maximum to be included in the best model to avoid over-fitting of the data.

Métier	No. of hauls	km.net hour Effort	No. of porpoises	Porpoise per haul	Porpoise per km.net hour
TN2	1356	52798	53	0.0391	0.0010

Table 19: Summary of harbour porpoise bycatch rates in tangle nets in ICES Area VII

Investigation of GAM plots indicated non-linear relationships between harbour porpoise bycatch rates and the explanatory variables fleet length, soak time, depth, latitude, longitude, month. A second order polynomial gave the same AIC as a linear term in the simple model and the AIC was not improved by using a categorical variable of fleet length (length cat A=<500 B=500-999 C=>1000). Although fleet length was not linear it could be approximated to linear and so a linear term was used.

The AIC for soak time in the simple model was not improved by a second or third order polynomial but improved when soak time was treated as a categorical variable (A=<24 hrs, B=24-48, C= >49). For the covariate depth, the lowest AIC value was achieved when the exploratory GLM was allowed to determine two slopes for this explanatory variable. Latitude was improved by a quadratic term. The AIC for month was the same for either a third order polynomial or the categorical variable "quarter" so quarter was used. Year (as factor) worked better than year as a categorical or continuous variable.

Variables were then investigated for collinearity. Year and rigged net height were co-aliased with mesh size and fleet length. This is likely a result of the limited variability in the gear characteristics of nets used in experimental trials. The final variables used in stepwise model selection are listed in Table 20.

Covariate	GVIF	Df
Fleet length	1.8	1
Soak time as categorical variable	1.2	2
Depth as quadratic term	2	2
Mesh size	1.4	1
Month as categorical variable	4.0	3
ICES subdivision	2.6	2
Latitude as quadratic	2.1	2
Longitude as quadratic	3.0	2
Presence or absence of floats	2.4	1

Table 20: GVIF scores for modelled covariates

The best model retained the variables fleet length, latitude and an interaction between fleet length and latitude. Although this only amounted to 5 variables the fitted values from the model, when harbour porpoise bycatch rates were predicted for low medium and high values of fleet length and latitude, exceeded those in the raw data and indicated a failure of the model to fit the data properly.

Step selection was rerun multiple times so that each term with a quadratic could be offered as a linear term and a term with two slopes. None of these changes in the form of any of the variables improved the AIC of the model or reduced the values of the parameter estimates.

All harbour porpoise bycatches occurred in ICES subdivision IVb where 87% of the observed effort (number of hauls) was recorded. Therefore to allow investigation of any spatial affect on bycatch rates the data were further sub-set to only those hauls that were observed in ICES subdivision IVb. Table 21 provides a summary of harbour porpoise bycatch rates per haul and km net hour for this subset.

No. of hauls	Effort (km net hour)	No. of porpoises	Porpoise per haul	Porpoise per km.net hour
1202	25966	53	0.044093	0.002041

Table 21: summary of observed hauls, effort and harbour porpoise bycatch rates for ICES subdivision IVb.

Assessment of GAM plots for the reduced data set showed a linear relationship between latitude and harbour porpoise bycatch rate. In a simple model the variable fleet length had a lower AIC when included as a categorical variable (A<399m, B:

400-999m, C$\geq$1000m). Depth gave the lowest AIC as a categorical variable and longitude was better when offered as two slopes (<= -0.01). Observer was co-aliased with year, and year had a GIF>10. The final model retained the variables fleet length and latitude (Table 22).

Covariate	Estimate	95% C.I.	P value
(Intercept)	158.2694	34.07 to 282.47	P<0.05
Fleet length: 400-999m	-1.8655	-2.56 to -1.18	P<0.001
Fleet length: $\geq$ 1000m	-1.8348	-2.47 to -1.19	P<0.001
Latitude	-3.0169	-8.27 to 2.23	P<0.01

Table 22: summary of parameter estimates, confidence intervals and p-values for the best model.

The model was rerun using a quasi-Poisson error distribution. The dispersion parameter φ was estimated at 1, indicating that the data were not over-dispersed. However, the large parameter estimate for the intercept resulted in high predicted bycatch rates at low, medium and high values of fleet length and latitude.

Of the 1,202 hauls observed in area IVb, 83% had been observed as part of experimental trials conducted to test the effects of different gear characteristics on bycatch rates. These experimental trials account for 46 of the 53 harbour porpoise caught in tangle nets in subdivision IVb. The significant relationship with latitude is driven by these experimental trials which were all conducted off the coast of Bridlington and the relationship with shorter fleet lengths is driven by the experimental trial conducted in 2003 where bycatch rates in standard skate nets and acoustically reflective (BaSO$_4$) tangle nets were investigated. When only experimental hauls were modelled (977 observed hauls) the only explanatory variable retained was a significant negative relationship between harbour porpoise bycatch rates and fleet lengths longer than 150m (p<0.001). This relationship is driven by the relatively high bycatch rates per km net hour that were observed in two experimental trials where the average fleet length of nets was between 91 metres and 114 km long (Table 23). Experimental trial A compared harbour porpoise bycatch rates in monofilament and multifilament nets, B compared harbour porpoise bycatch rates in thick and thin twinned nets, C compared harbour porpoise bycatch rates in standard and BaSO$_4$ nets and D compared harbour porpoise bycatch rates in standard rigged skate nets and nets rigged with double the amount of standard flotation.

Experimental trial	Number of hauls	Average fleet length	Effort (km net hours)	Number of Porpoises	Bycatch rate (by haul)	Bycatch rate (km net hours)
A	181	114	1057	10	0.06	0.009
B	203	1000	9600	14	0.07	0.001
C	327	91	1305	11	0.03	0.008
D	286	640	7168	11	0.04	0.002

Table 23: summary of harbour porpoise bycatch rates and fleet lengths in observed tangle net hauls during 4 experimental trials.

Figure 12 shows the mean values of soak time and depths for tangle nets during these four experimental trials. All experimental nets, except for $BaSO_4$ nets, had a mesh size of 267mm, which is standard in UK tangle net fisheries targeting skate.
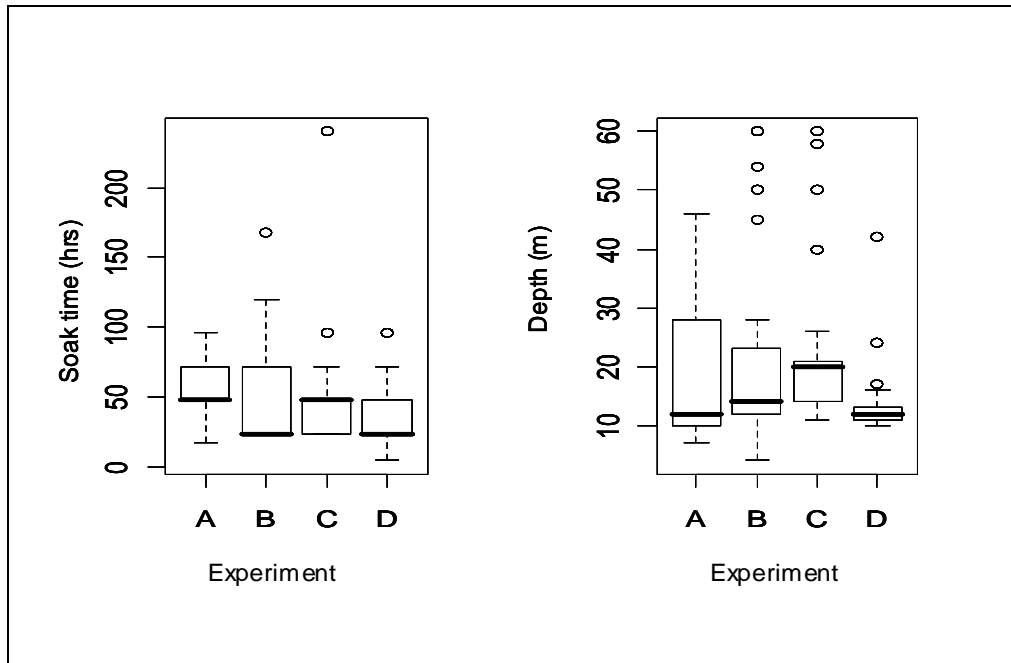
Figure 12: box plots of soak time and depth for tangle nets observed in four experimental trials.

In trial C, no significant difference in bycatch rates was observed between standard skate nets and $BaSO_4$ nets (Northridge et al. 2003), however $BaSO_4$ nets were taller than the standard skate nets and also had a smaller mesh and thicker twine diameter. Investigation of residuals from the final model showed that bycatches were not significantly correlated in subsequent huals (Fig. 13).

Fig 13: autocorrelation of bycatch events in subsequent hauls within trips. Red dots indicate 95% confidence intervals generated after re-sampling the final model 100 times. Horizontal lines show the correlation at each lag.

2.4.6.5 Harbour porpoise bycatch in Area VII gillnets

Once missing values had been removed, the final data set available to model for gillnets in ICES area VII consisted of 2,110 hauls and 24 bycaught harbour porpoise. To avoid over-parameterisation, a maximum of four parameters were allowed in the best model. All harbour porpoise bycatches were recorded in gillnets targeting cod and other whitefish (GN2) and gillnets targeting hake (GN3) (Table 24).

Métier	No. of hauls	Effort	No. of porpoises	Porpoise per haul	Porpoise per km.net hour
GN1	215	1572	0	0	0
GN2	926	13187	9	0.0097	0.0007
GN3	345	40560	15	0.0435	0.0004
GN5	2	3	0	0	0
GN6	508	10473	0	0	0
GN7	114	258	0	0	0

Table 24: Summary of harbour porpoise bycatch rates by gillnet métier in ICES Area VII.

The relationship between explanatory variables and harbour porpoise bycatch rates was investigated visually using GAM plots. Fleet length, longitude, depth and height all had non-linear relationships with harbour porpoise bycatch rates. Longitude had a

lower AIC in the simple model when offered as a quadratic term, and depth had a lower AIC when offered as a categorical variable ($\leq$ 56m and >56m). Both fleet length and net height also had a lower AIC in the simple model when treated as a categorical variable. However, given the low number of harbour porpoises bycaught in this dataset, to prevent over-parameterisation both covariates were turned into dummy variables. Therefore total fleet length was treated as three dummy variables (<300m, <550m, >1500m) and net height was treated as four dummy variables (<2m, $\leq$4m, $\leq$ 5.9m and >6 m).

All possible explanatory variables were tested for collinearity. Table 25 shows the final variables used in forward backward model selection. LOA and observer were co-aliased coefficients, and ICES sub-division, year (as a categorical variable) and longitude all had a GIF >10.

Variable	GVIF	Df
Fleet length as categorical variable	5.75	3
log(soak time)	1.35	1
Mesh size	3.96	1
Absence or presence of floats	2.94	1
Latitude	1.43	1
Depth (<= 56m, >56m)	1.49	2
Month	1.60	1
Net height as categorical variable	6.55	3

Table 25: GVIF scores for modelled covariates

The best model retained the variables height ($\leq$4m), latitude, soak time, height ($\leq$5.9m) mesh and depth, and an interaction between latitude and soak time, and latitude and mesh. As the maximum number of variables acceptable for this model was four, step wise selection was rerun, removing the variable that increased AIC the least at each step until a final model with 4 or less explanatory variables was produced. The final model after this process retained the variables height ($\leq$4m), latitude, mesh size and an interaction between mesh size and latitude. Table 26 summarises the outputs of this model.

Covariate	Estimate	95% C.I	p-value
(Intercept)	-1257	-1957.8 to -556.2	p<0.001
height <= 4	-35.22	-2905.22 to 2834.78	p>0.1
Latitude	24.73	10.79 to 38.67	p<0.001
mesh	9.06	3.8 to 14.32	p<0.001
Latitude: mesh	-0.18	-0.28 to -0.08	p<0.001

Table 26: summary of parameter estimates, confidence intervals and p-values for the best model.

The standard errors around the estimate for the height variable are extremely large and predicted values of harbour porpoise bycatch from this model at low, medium and high values of each explanatory variable exceeded those in the raw data. In particular predictions with latitude did not fit well.

Therefore step selection was rerun, this time omitting the variable latitude. The best model, after reduced steps to allow a maximum of four variables, retained the covariates height ($\leq$ 4m), height (<6m), mesh size and depth. The parameter estimate and standard errors for height < 4m were again very large (Table 27).

Covariate	Estimate	95% C.I	p-value
(Intercept)	-15.94	-21.54 to -10.37	p<0.001
height $\leq$ 5.9m	2.25	-0.11 to 4.4	p<0.05
height $\leq$ 4m	-15.76	-2039.8 to 2008.2	p>0.1
mesh	0.06	0.02 to 0.1	P<0.01
depth	-0.01	-0.05 to 0.03	P<0.1

Table 27: Summary of parameter estimates, confidence intervals and p-values for the best model.

These results indicate that while nets between 4.1m and 6m have a significantly higher bycatch rate than nets taller than 6m, there is little information given by the model about the relationship between nets less than 4m in height and harbour porpoise bycatch. While this model does not provide a useful parameter estimate for nets with total heights less than 4m, the predicted values from this model were much improved.

Figure 14 shows the relationship between each net height category and mesh size and depth.

Figure 14: Box plots of depth fished and mesh size for each height category of gillnets observed in ICES Area VII.
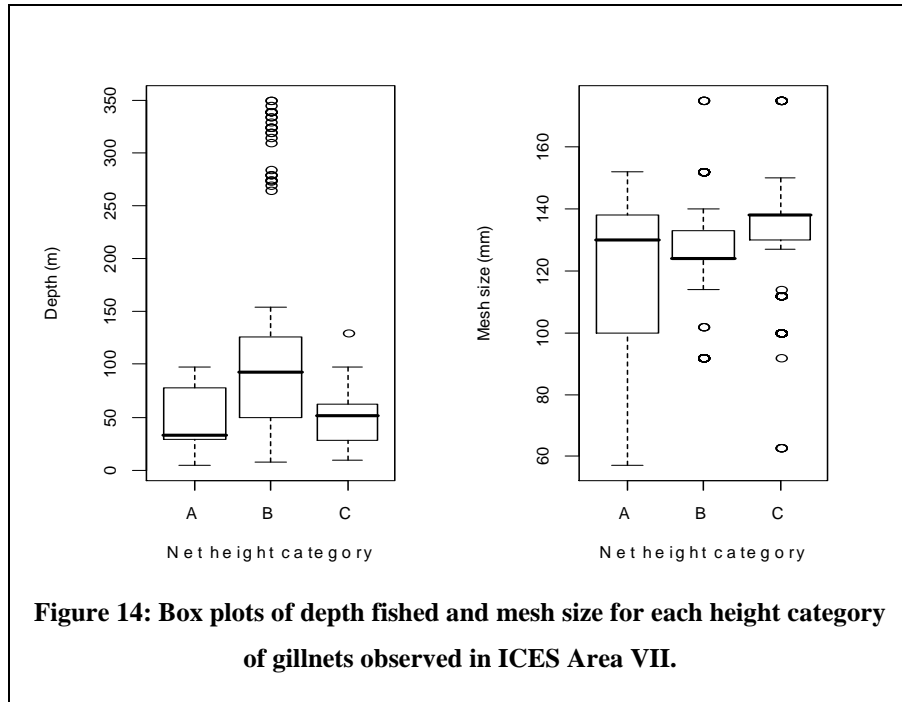
The model was rerun with a quasi-Poisson error distribution. The dispersion parameter Φ was estimated as 1.7. Therefore the final model was compared to the same model except with negative binomial error distribution using the package pscl. Results showed no reason to reject the null hypothesis that the errors followed a Poisson distribution (p-value = 0.1659). Figure 15 shows predictions of harbour porpoise bycatch rates at interim values of net height, depth and mesh size.
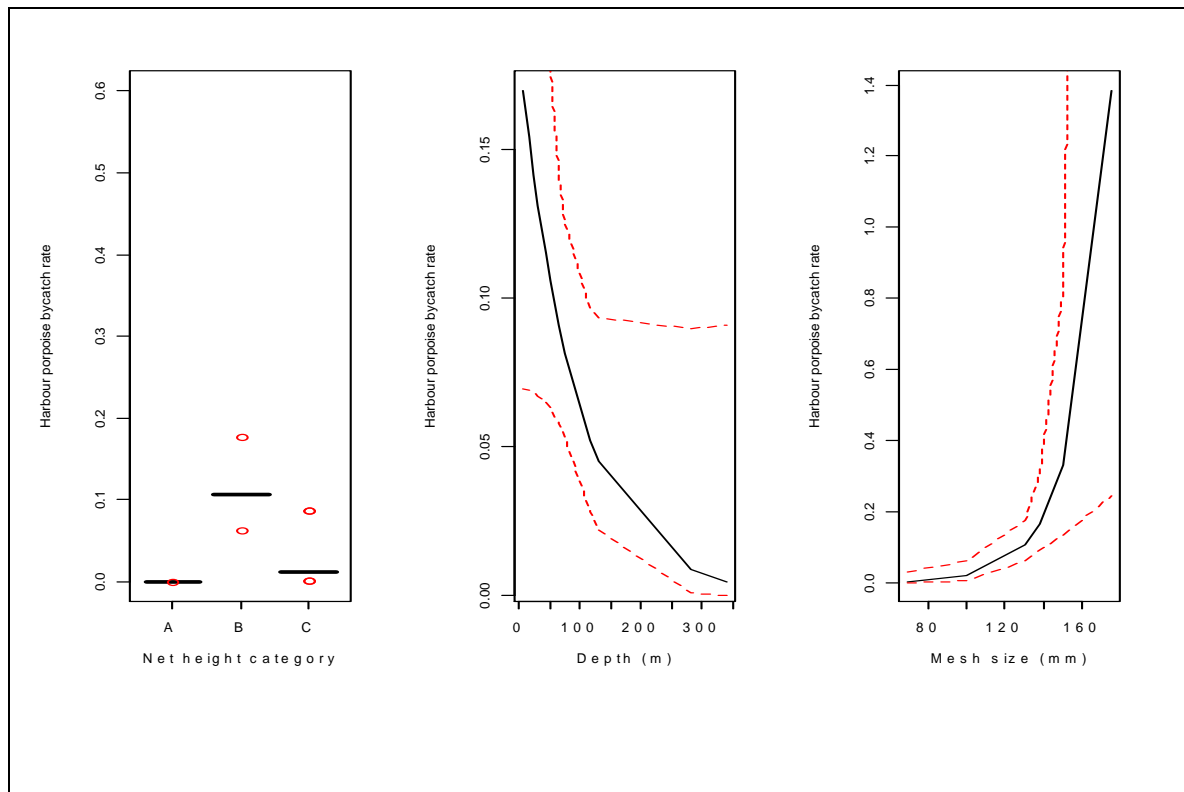
Figure 15: Plots showing the relationship between predictor variables and harbour porpoise bycatch (offset by km.net hour). The red dashed lines indicate the 95% confidence intervals. N.B. y-axis scales are different.

Investigation of the residuals of this model showed bycatch events were significantly autocorrelated at a lag of 1 (Figure 16). This is not surprising given that of the 17 trips where harbour porpoise bycatch was observed, 3 trips had more than 1 haul with bycatch recorded. Of these 3 trips, 1 trip caught a single animal in two separate hauls, 1 trip caught a single animal and two animals in two separate hauls, and 1 trip caught 5 animals in 4 separate hauls (1 haul with two animals).
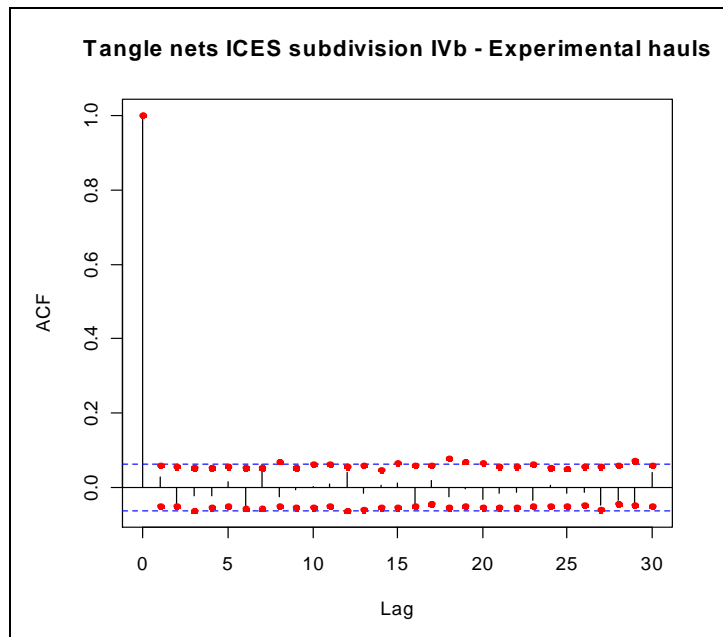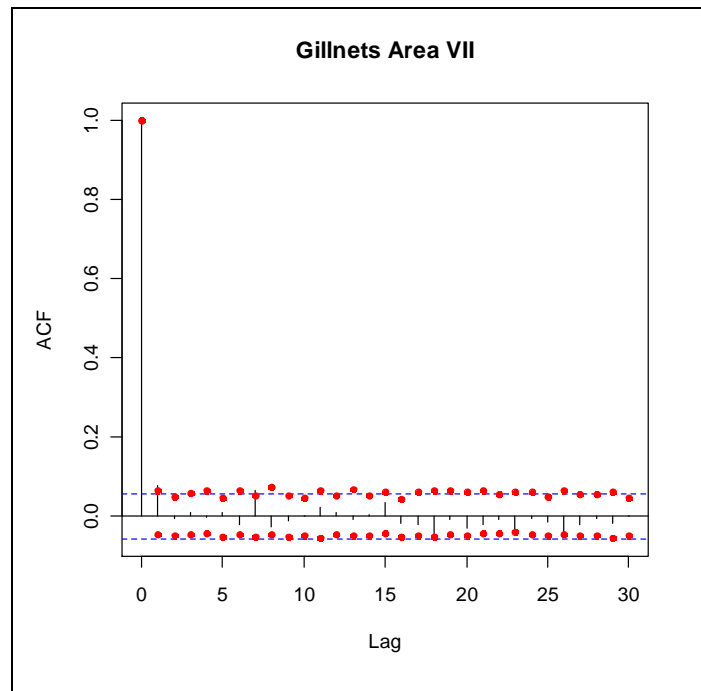
Fig 16: Autocorrelation of bycatch events in subsequent hauls within trips. Red dots indicate 95% confidence intervals generated after re-sampling the final model 100 times. Horizontal lines show the correlation at each lag.

Due to the very large parameter estimates given for the height category <=4m the data set was re-sampled 10 times and step selection was rerun on each of the 10 re-sample data sets. The best model selected by AIC varied between interactions indicating the model was unstable. However, of these 10 replicates, all retained the two height variables, 90% retained the variable mesh size and 70% retained the variable depth and all had consistent parameter values for the shared covariates. The model instability is likely an artefact of small sample size but the consistency in retained explanatory variables across the replicates suggest that these relationships are not purely the result of chance effects on a small sample size. The addition of further data would allow these relationships to be confirmed.

As all bycatches were recorded in métiers GN2 and GN3 these data were then modelled separately. Although these nets are characterised as either short wreck nets for cod, and longer nets targeting cod or hake, the model did not retain fleet length. This is a result of similar bycatch rates, per km net hour, between the two métiers (Table 28). The best model retained all the same variables retained in the model of the larger dataset, except depth.

Fleet length (m)	Total effort (km.net hrs)	Number of hauls	No. Of Porpoises	Bycatch rate per km.net hour	Bycatch rate per haul
<=760	6020	835	4	0.000664	0.00479
>760	47787	437	20	0.000419	0.04577

Table 28:Summary of harbour porpoise bycatch rates in short gillnets targeting cod and longer gillnets targeting cod and other whitefish and hake.

Figure 17 summarises the total height of nets and mesh sizes used in the two métiers, for fleet lengths of 760m or less, or fleet lengths longer than 760m. Shorter wreck nets had a higher average net height, however, the wide range of heights recorded for these nets, and the outliers recorded for longer hake and cod nets, indicate that there are likely to be inaccuracies in some of the net heights recorded by observers. Observers do not have the opportunity to directly measure net height onboard and must therefore rely on information provided by skippers.
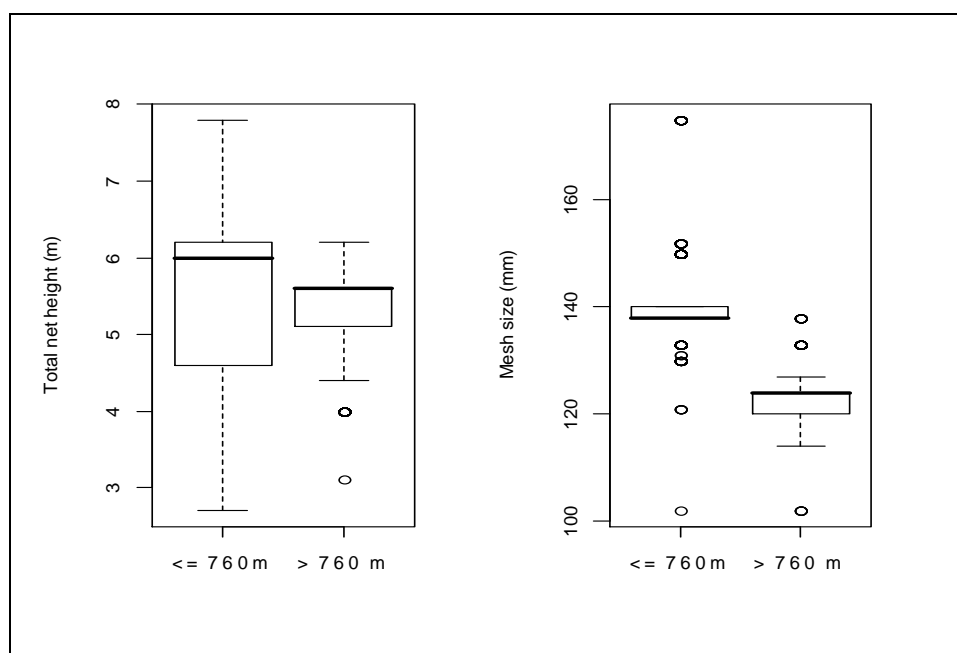
Fig 17: Box plot of total net height and mesh size for short nets targeting cod and longer nets targeting cod and hake in ICES Area VII.

2.4.6.6 Harbour porpoise bycatch in Area VII tangle nets

Once missing values were removed, the final data set modelled for tangle nets in ICES area VII consisted of 1,645 hauls and 19 bycaught harbour porpoise. Therefore

a maximum of 3 parameters were allowed in the best model. 94% of hauls and all harbour porpoise bycatches were observed in tangle net métier TN2 (Table 29).

Métier	No. of hauls	Effort	No. of porpoises	Porpoise per haul	Porpoise per km.net hour
TN1	8	1063.7	0	0.0000	0.0000
TN2	1548	296530.9	19	0.0123	0.0001
TN3	89	10257.5	0	0.0000	0.0000

Table 29: Summary of harbour porpoise bycatch rates by tangle net métier in ICES Area VII.

Inspection of GAM plots showed linear relationships between harbour porpoise bycatch rates and soak time, latitude, depth, year and month. Fleet length, mesh size and longitude produced lower AICs in simple models when treated as categorical variables. However, to prevent over-parameterisation in the final model the simple model was allowed to choose two slopes for longitude ($\geq$ or < 4.5 degrees west) and mesh size (> or $\leq$270mm). Total fleet length was treated as three dummy variables (>999m, $\geq$100m and $\geq$200m). Observer and LOA were co-aliased, while subdivision had a GVIF>10. The final explanatory variables included in step selection are listed in Table 30.

Covariate	GVIF	Df
Fleet length as categorical variable	2.47	3
log(soak time)	1.35	1
depth	2.77	1
Latitude	1.36	1
Longitude with two slopes	1.00	1
Mesh with two slopes	2.94	1
month	1.08	1
year	1.49	1
Floats	1.00	1

Table 30: GVIF scores for modelled covariates

The best model using step selection retained the variables floats present, year, fleet length $\geq$ 1000m and fleet length $\geq$ 2000m (AIC = 186.9). To obtain no more than 3 parameters the step selection was rerun omitting fleet length >999m which added the least to AIC. The final model retained the parameters floats present, year and fleet length $\geq$2000m. However, parameter estimates and confidence intervals for both the intercept and floats present were extremely large (Table 31).

Covariate	Parameter estimate	95% Confidence interval	P-value
(Intercept)	-823.17	-1417.91 to -228.43	P<0.01
Floats present	-17.11	-2260.65 to 2226.43	P<1
Year	0.41	0.11 to 0.71	P<0.01
Fleet length >2000	-1.09	-2.05 to -0.13	P<0.5

Table 31: Summary of parameter estimates, confidence intervals and p-values for the best model.

The data were then re-sampled 10 times, with replacement, and stepwise model selection was conducted on each of the re-sampled data sets. The final model proved to be unstable. However, the covariate year was retained in 90% of the models and floats present was retained in 80% of the models. The category of fleet length retained was not stable during model selection. While floats and mesh did not show collinearity when tested, it is likely that these covariates are proxies for each other (Figure 18). No bycatches were observed in mesh sizes greater than 280mm and the majority of nets where floats were recorded present had mesh sizes of 300mm or more. However, 65% of all hauls were observed in nets with mesh sizes between 262 and 279mm. Given the relatively low number of harbour porpoise bycatches to model and the instability of the model, the retention of the covariate floats present may simply reflect a better fit of this categorical variable than mesh size to the data.
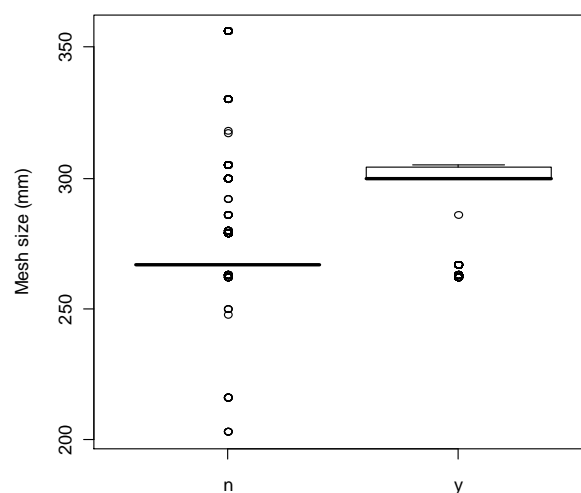
Figure 18: Box plot of mesh sizes in tangle nets with no floats present (n) and tangle nets with floats present (y).

2.4.6.7 Dolphin bycatch in all nets in Area VII

Twenty-seven dolphins were recorded bycaught in 3,709 hauls observed between 2004 and 2009. Of these, two individuals were bottlenose dolphins and the remaining animals were common dolphins. The bottlenose dolphins were observed in two separate trips, the first was caught in a tangle net targeting monkfish, the second was caught in a short gill net (180m) targeting Pollock. Given differences in the distribution and behavioural ecology of bottlenose dolphins and common dolphins, the two hauls with bottlenose dolphin bycatches were removed from the data prior to modelling. Therefore the model is explicitly capturing the relationship between covariates and common dolphin bycatch rates. Table 32 summarises bycatch rates of this species by métier in ICES Area VII. Highest bycatch rates per km net hour were recorded in tangle nets targeting crustaceans (TN3) followed by gillnets targeting cod and other whitefish (GN2).

Métier	No. of hauls	Effort	No. of Dolphins	Dolphins per haul	Dolphins per km.net hour
GN1	202	1473	0	0.000	0
GN2	860	12624	2	0.002	0.00016
GN3	345	40560	6	0.017	0.00015
GN5	2	3	0	0.000	0
GN6	479	9244	0	0.000	0
GN7	116	258	0	0.000	0
TN1	8	1064	0	0.000	0
TN2	1411	276168	8	0.006	0.00003
TN3	81	8172	9	0.111	0.00110

Table 32: Summary of common dolphin bycatch rates by métier in ICES Area VII.

No dolphin bycatches were observed in ICES subdivisions VIIa or VIId so these hauls were removed prior to analysis (n= 205). The final data set modelled comprised 3,504 hauls and 25 common dolphins.

Fleet length, soak time and depth showed linear relationships with common dolphin bycatch rates, while mesh size and month produced a lower AIC in the simple model when offered as categorical variables. Latitude provided a better fit to the data when offered as a quadratic term, however, longitude also had a non-linear relationship with bycatch but the shape of this relationship was difficult to capture. Subdivision and

longitude were collinear, therefore subdivision was retained instead of longitude, as a suitable form of the latter covariate could not be determined. LOA and observer ID also had a GVIF>10. Table 33 shows the final covariates used in stepwise selection. To avoid over-parameterisation a maximum of five explanatory variables were allowed to be retained by the final model.

Covariate	GVIF	Df
log(fleet length)	4.7	1
log(soak time)	3.2	1
depth	3.0	1
Latitude as quadratic term	3.1	2
Mesh size as categorical variable	7.4	4
Month as categorical variable	3.8	3
subdivision	7.8	4
as.factor(year)	3.8	5

Table 33: GVIF scores for modelled covariates

The covariates mesh size and month were treated as dummy variables to prevent over parameterisation of the model. The dummy variables for mesh size were <100mm, <138mm, <203mm and $\geq$ 203mm. The dummy variables for month were January-March, April-June, July-September and October-December. The best model retained the variables soak time, the mesh category $\geq$ 203mm , and month category October – December, and an interaction between the mesh category and soak time and the month category and soak time (Table 34).

Covariate	Estimate	95% Confidence interval	P-value
(Intercept)	-14.2	-20.36 to -8.02	p<0.001
log(soak time)	1.1	-0.49 to 2.7	p>0.1
Mesh $\geq$ 203mm	10.9	7.19 to 14.62	p<0.001
Month (Oct – December)	10.4	4.37 to 16.52	p<0.001
log(soak time): Mesh $\geq$ 203mm	-2.7	-3.82 to -1.53	p<0.001
log(soak time): Month (Oct – December)	-2.4	-3.98 to -0.92	p<0.01

Table 34: Summary of parameter estimates, confidence intervals and p-values for the best model.

Figure 19 shows predictions of common dolphin bycatch rates at interim values of mesh size (124mm) and soak time (24 hours).
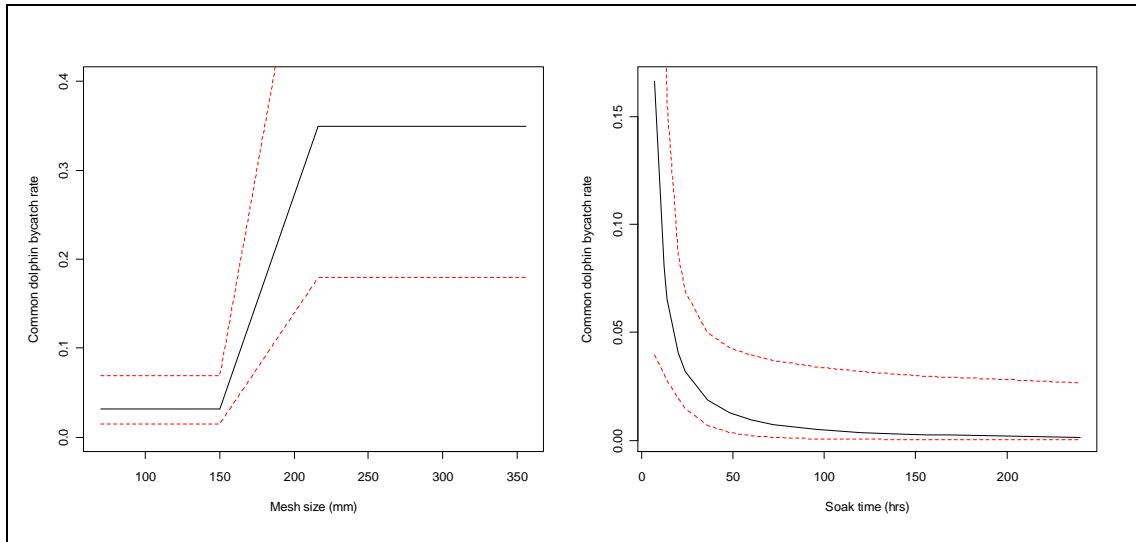
Figure 19: Plots showing the relationship between predictor variables and common dolphin bycatch (offset by km.net hour). The red dashed lines indicate the 95% confidence intervals. N.B .y-axis scales are different.

Investigation of the residuals of this model showed high autocorrelation of bycatch rates between hauls (Figure 20). This autocorrelation is in part driven by 1 trip where a total of 8 dolphins were caught in 3 hauls. This trip was removed and the data were remodelled as a check to see if the relationships were upheld. The best model retained, when this trip was removed, retained the variables soak time, month category (October – December) and an interaction between month and soak time. The relationship between common dolphin bycatch and soak time interacting with month category remained negative for this model. This is likely due to the clumped nature of bycatch events in space and time.
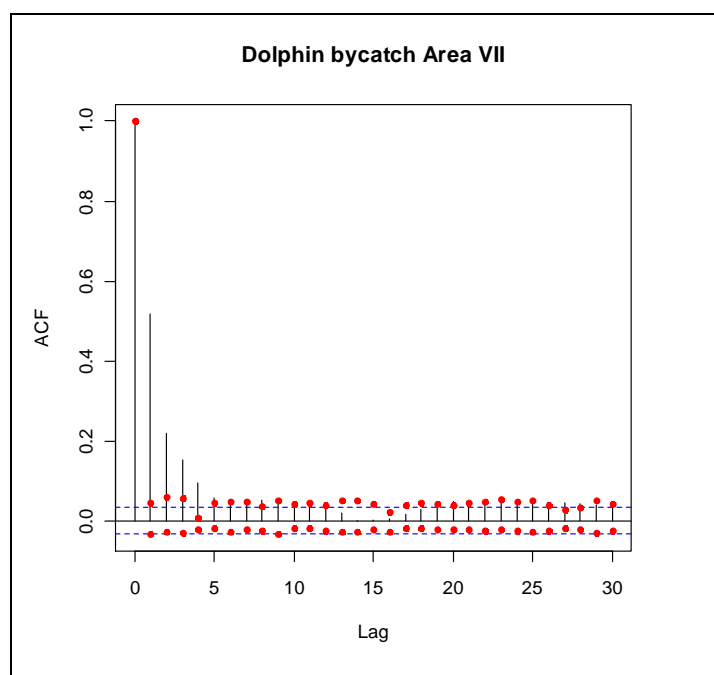
Fig 20: autocorrelation of bycatch events in subsequent hauls within trips. Red dots indicate 95% confidence intervals generated after re-sampling the final model 100 times. Horizontal lines show the correlation at each lag.

Running the autocorrelation function on this model showed data were no longer auto correlated at lag 1, although they were significantly correlated at lag 5. Further investigation of the dataset revealed that 11 of the 25 common dolphin bycatches were recorded on the same boat, in four successive trips observed over a period of 8 days. This vessel fished both wreck nets targeting whitefish and tangle nets targeting monkfish and spider crabs. During the first trip, where only wreck nets were hauled, one common dolphin was bycaught. During the 2nd trip one dolphin was caught in a wreck net, but none were caught in tangle nets hauled, and during the 3rd trip, a total of eight dolphins were caught in three tangle nets hauls. During the 4th trip, one dolphin was caught in a tangle net. Therefore model results are driven by the four trips observed on this vessel. The tangle nets used by this vessel were 1125m long while the wreck nets were between 285 and 752m long. Mesh size of wreck nets were 130mm and tangle nets were 300mm. During these trips both net types were soaked for 12 hours prior to hauling, which is a relatively short amount of time. Bycatch rates per month are summarised in Figure 21. The eight animals observed bycaught in the same trip drive the high bycatch rates in November.

Figure 21: common dolphin bycatch rates per month.
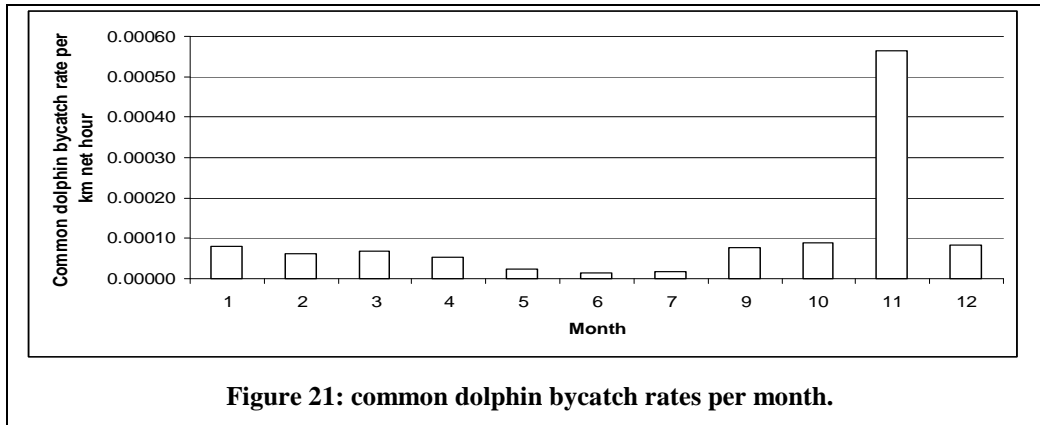
2.4.6.8 Relationship between harbour porpoise bycatch rates and fish catches

Data related to the amount of fish catch in kg were only available for 4,145 hauls. Remaining hauls had fish catch recorded, as number of individuals, and conversion factors were unavailable to convert these data to weights. To assess the relationship between total fish catch and harbour porpoise bycatch rates, only those hauls where fish catch had been recorded could be analysed. Therefore, 120 hauls where no fish catch was recorded were removed from the dataset, in which one harbour porpoise had been recorded bycaught. GAMs with a quasi-Poisson error distribution were constructed for three combinations of the data. The quasi-Poisson error distribution was chosen to account for over dispersion in the data. Figure 22 shows the relationship between harbour porpoise bycatch rates and logged fish catches for gillnets and tangle nets combined, and for both gillnets and tangle nets analysed separately. For all three combinations of the data harbour porpoise bycatch rates initially increased with the amount of fish caught. However, in gillnets after this initial increase, there was a decrease in bycatch rates, which was then followed by an increase. For tangle nets, after the initial increase, bycatch rates levelled out at fish catches around 400kg per haul. The effect of different measures of effort on fish catches in gillnets was then investigated using a GAM with quasi-Poisson error distribution. Figure 23 shows the relationship between fish catches and soak time, fleet length and effort. While fish catches generally increased with soak time, there was a peak in catch in kg at fleet lengths of approximately 250m (wreck nets), before a decrease in catch and then a linear increase as fleet length increased.
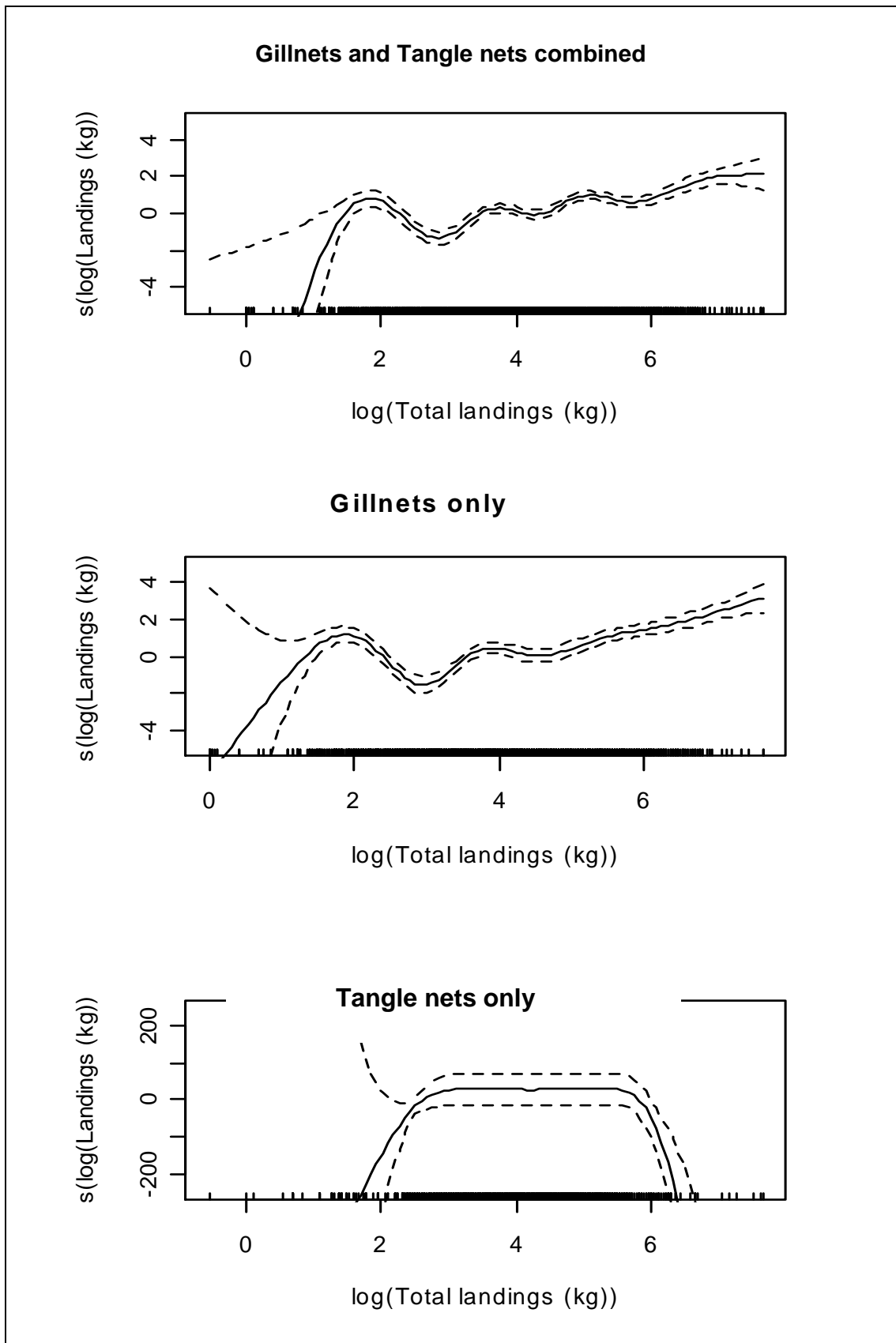
Figure 22: Relationship between harbour porpoise bycatch rates and total fish landings for all nets combined, gillnets only and tangle nets only. These plots are on the linear predictor scale and show the smooth of log total fish landings centred on zero. Therefore they show the shape of the relationship with harbour porpoise bycatch rates rather than absolute values.
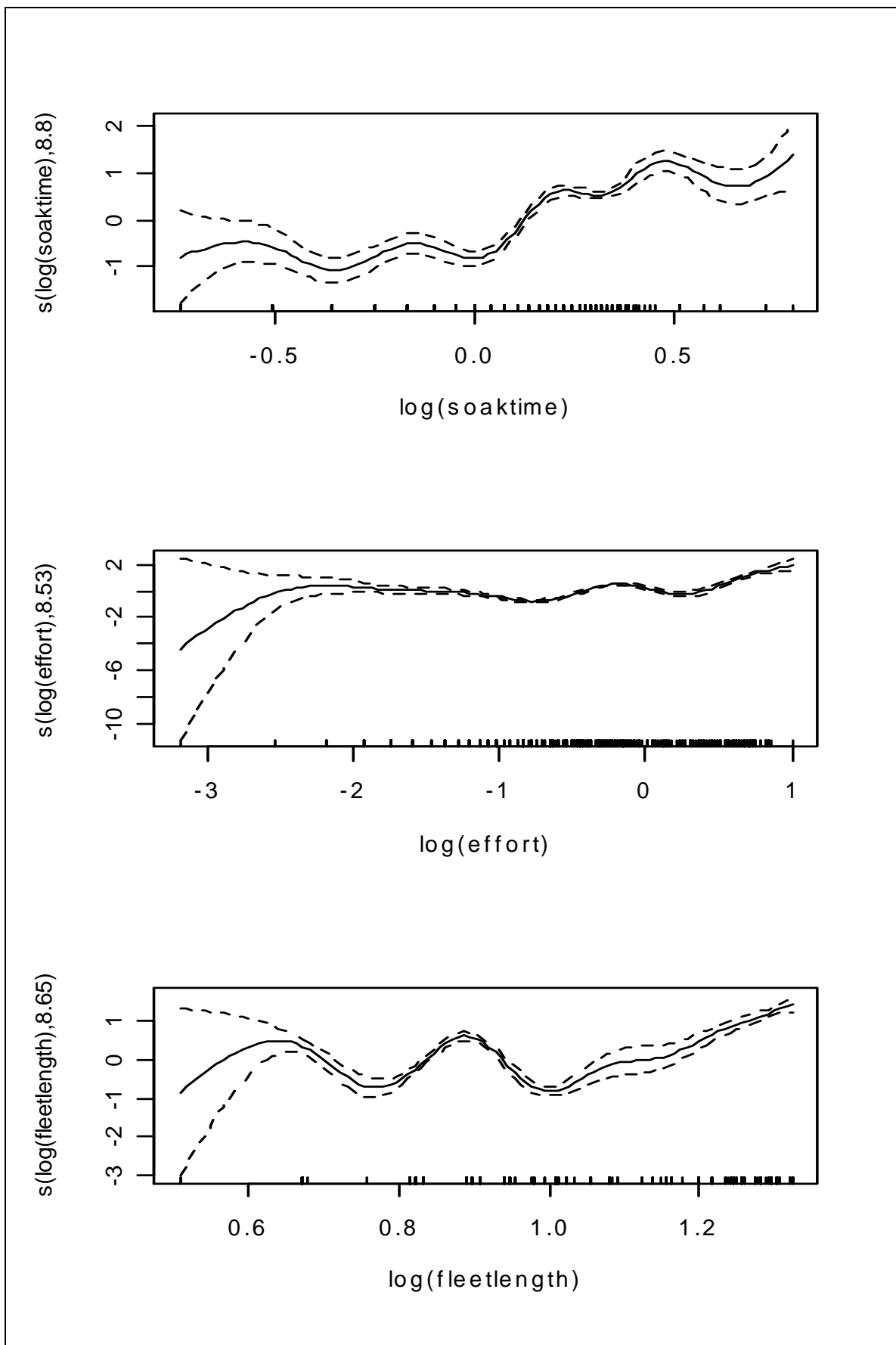
Figure 23: Relationship between harbour porpoise bycatch rates and total fish catches in gillnets, using different metrics of fishing effort. These plots are on the linear predictor scale and show the smooth of the log of each variable centred on zero. Therefore they show the shape of the relationship with harbour porpoise bycatch rates rather than absolute values.

2.5 Discussion

2.5.1 Factors affecting harbour porpoise bycatch rates in gillnets and tangle nets.

When independent onboard observer data collected in gillnet and tangle net hauls were combined for each ICES area the significant explanatory variables retained for the best models to predict harbour porpoise bycatch rates were fleet length (ICES area IV) and rigged net height (ICES area VII). The significant negative relationship between harbour porpoise bycatch and fleet length in ICES area IV was driven by highest observed bycatch rates occurring in short gillnets targeting cod (<500m) and short tangle nets (<150m). The significant positive relationship between harbour porpoise bycatch and net height in ICES area VII was driven by highest observed bycatch rates occurring in short wreck nets (GN2) and in long gillnets targeting hake (GN3) both of which had an average rigged net height of 5.2m. While the results of these models identified specific fisheries with high bycatch rates, they did not provide information on which characteristics within these, and other fisheries, might be appropriate to investigate for their potential to mitigate bycatch. For this reason separate models were constructed to investigate bycatch rates of harbour porpoises in gillnets and tangle nets for ICES Area IV and VII. The results of these models and utility of retained explanatory variables in relation to potential gear modifications are discussed below.

2.5.2 Fleet length

Fleet length was found to have a significant negative relationship with harbour porpoise bycatch rates in the best models retained for both gillnets and tangle nets in ICES Area IV. In this area, 95% of harbour porpoise bycatches were recorded in gillnets targeting cod and other whitefish, prior to 1999. This métier (GN2) accounted for 87% of all hauls observed in ICES Area IV, and 16 of the 19 porpoises observed in this métier were caught in nets less than 500m in length, the remaining 3 in nets less than 1000m in length. While no porpoises were observed bycaught in nets longer than 1000m, these nets only accounted for 2.6% of all observed hauls. These short fleets of net, mostly targeting cod, are known as wreck nets. Wreck nets are similar to

standard cod nets, although with slightly larger mesh sizes, and are shot over wrecks or rough ground to target aggregations of fish. Vinther (1999) also reported that the bycatch rates of harbour porpoise were higher in Danish North Sea wreck nets than in longer cod nets. The negative relationship between fleet length and harbour porpoise bycatch rates in the North Sea described in both this study and by Vinther (1999) indicate that some other characteristic of wreck net fisheries results in increased bycatch rates. This may be due to higher densities of harbour porpoise prey species around wrecks or the close proximity that wreck nets are set together in an area.

The best model for predicting harbour porpoise bycatch rates in tangle nets in ICES Area IV was for a subset of the data which contained those hauls observed in ICES subdivision IVb. 83% of the observed hauls in this subdivision were part of experimental trials testing bycatch rates in nets with different gear characteristics, which were conducted in the waters off Bridlington, North Yorkshire. The explanatory variables retained by this model were fleet length, as a categorical variable, and latitude. Bycatch rates, per km net hour, were significantly higher in fleet length less than 150m, and this relationship was driven by two experiments in years 2000-2001 and in 2003. While soak time was not retained as an explanatory variable in the best model, average soak times were highest in the two aforementioned trials compared to the two experimental trials using longer fleet lengths.

In Area VII all harbour porpoise bycatches were recorded in gillnets targeting cod and other whitefish (GN2), or in gillnets targeting hake (GN3). These metiers accounted for 60% of the total number of observed hauls in this area. Bycatch rates, when calculated by km net hours, were reasonably similar between wreck nets (<760m) and cod and hake nets longer than 760m in length (0.007 to 0.004 respectively). Therefore, in contrast to results from Area IV, fleet length was not retained as a significant predictor of harbour porpoise bycatch in the final model. Instead, the best model for this area indicated a significant relationship between harbour porpoise bycatch rates and rigged net height. The length of hake nets fished in ICES Area VII has increased in the last decade as fish catches have decreased (Northridge & Mackay 2004). Tregenza reported a bycatch rate of 7.7 harbour porpoise per 10,000 km net hours observed in the Celtic Sea between 1992 and 1994, when nets were reported to generally be 1.6km in net ("although with much variation"). 343 hauls have been

recorded in UK hake net fisheries since 2004, the average net length is now 4.8km and using the same metric as Tregenza *et al.* (1997), the bycatch rate of harbour porpoise is 3.4 animals per 10,000 km hours. Therefore an increase in net length in this fishery has not resulted in an increase in harbour porpoise bycatch rates.

While harbour porpoise bycatch rates per haul were the same for wreck nets in ICES Area IV and VII (0.01 animals per haul), bycatch rates per km net hour were much higher in wreck nets in ICES Area IV (0.02 v. 0.007). This higher bycatch rate per km net hour is a result of the shorter average fleet length and shorter average soak durations of wreck nets in ICES Area IV compared to Area VII.

The best model of harbour porpoise bycatch in tangle nets in ICES Area VII retained the explanatory variables floats present, year and fleet lengths ≥ 2000m. However, the parameter estimates for the presence of floats was large, and when re-sampled the model proved to be unstable. Nonetheless, a positive relationship with year was retained in 90% of the re-sampled models, whilst 80% retained a negative relationship with the presence of floats. A negative relationship with fleet lengths ≥ 2000m was only retained in 30% of the models, indicating that it is unlikely that this variable has a strong relationship with harbour porpoise bycatch rates.

In contrast to the finding of a significant negative relationship between harbour porpoise bycatch rates in gillnets and tangle nets in ICES Area IV, both Palka (2000) and Orphanides (2009) report a significant positive relationship between harbour porpoise bycatch rates and fleet length in USA mid-Atlantic sink gillnet fisheries. Both these studies combined data from gillnets and tangle nets fisheries. The significant negative relationship with fleet length found in the UK observer data is likely driven by the high proportion of observer coverage in short wreck net fisheries and the inclusion of experimental tangle net hauls observed in ICES Area IV. 87% of observed gillnet hauls and 86% of observed tangle net hauls observed in this area were in fleets less than 1000m. As harbour porpoise bycatch in UK static net fisheries was modelled as a rate, using an offset of km net hours per haul, the retention of the explanatory variable fleet length indicates that the assumption of a proportional relationship between harbour porpoise bycatch and fishing effort does not always hold true. Palka (2000) modelled bycatch per haul, while Orphanides (2009) used the total

catch landed as a measure of fishing effort. Vinther (1999) calculated highest bycatch rates in Danish cod fisheries when using km net hours as a measure of effort. However, when bycatch rates were modelled using fleet length as a measure of effort the highest rates were in the Danish turbot fishery. The results of these studies show that the apparent relationship between harbour porpoise bycatch rates and fleet lengths can be affected by the measure of fishing effort used.

2.5.3   Net height

The positive significant relationship between rigged net height and harbour porpoise bycatch rates in ICES Area VII is driven by high bycatch rates in gillnets targeting cod and other whitefish (GN2) and gillnets targeting hake (GN3). Although the best model proved unstable when re-sampled, this instability is likely a result of a relatively small number of bycatch events in relation to the total area observed, and the variability of gear types sampled. Re-sampling with replacement can limit the number of total observed bycatch events available to the model and therefore the number of total variables that should be retained to avoid over-parameterisation. However, the retention of the net height variable in 100% of the re-sampled models, even with replacement, suggest that this relationship described by the model has not occurred simply by chance. The mean rigged height of gillnets targeting cod and gillnets targeting hake in ICES Area VII was 5.5m and 5.2m respectively. These métiers accounted for 60% of all observed hauls in gillnet fisheries in this area. The next most frequently sampled gillnet métier in ICES Area VII was sole nets (GN6) which accounted for 24% of observed hauls during which no harbour porpoises were observed bycaught. While the mesh size of sole nets is similar to those used in gillnets targeting cod and hake, the rigged height of these nets is much lower, averaging 1.6m. Sole nets are also constructed of thinner nylon twine than either gillnets for cod or hake. Although UK observers record twine diameter when possible this variable had too many missing values to be included as a covariate in the model. However, available data show that the average twine diameter of sole nets is 0.35mm compared to 0.64mm for nets targeting hake. It is unclear whether the lower profile, or the thinner netting material of sole nets (or some other factor) resulted in the lack of harbour porpoise bycatch observed in this metier. In contrast rigged net height was not retained as a significant predictor of harbour porpoise bycatch in gillnets in ICES

Area IV. However, 87% of observed gillnet hauls in this area were in wreck nets and longer cod nets, both of which had an average rigged height of 3.6m.

The effect of rigged net height on harbour porpoise bycatch rates could not be investigated for tangle nets in ICES Area IVb as the dataset was dominated by hauls observed during experimental trials where nets either had a rigged height of 2m or 4m, which resulted in rigged net height being co-aliased with the explanatory variables mesh size and year. During one experiment that directly compared bycatch rates in nets with a rigged height of 2m and 4m respectively, bycatch rates per haul were higher in the 4m nets, but the difference was only significant at the 10% level (Northridge et al 2003). However, the mesh size and twine diameter of these two net types was also different.

Due to the instability of the final tangle net model for Area VII it is not possible to draw conclusions on the effect of net height on harbour porpoise bycatch rates. While the model indicated a negative relationship between harbour porpoise bycatch rates and the presences of floats, which should increase the fishing profile of nets, it is likely this explanatory variable was acting as a proxy for mesh size.

Although the effect of rigged net height on cetacean bycatch rates has not been directly investigated, the use of tie downs in large mesh USA Atlantic sink net fisheries were found to be associated with lower bycatch rates of harbour porpoises (Palka 2000) and their use is now mandatory in some fisheries. Tie downs are lines that are shorter than the height of the fishing net and connect the float line to the lead line of the net at equal distances along the net. By using tie downs, not only is the height of the net reduced, but also the meshes of the net form a bag that aid in the entanglement of demersal fish. Preliminary results of a recent experimental trial to investigate the effect of tie downs on bycatch rates of Atlantic sturgeon, found catch rates of sturgeon were lower in nets without tie downs. However, common dolphins bycatches also occurred in these nets, while none were recorded in control nets with tie downs (ICES 2011). This result suggests that lower profile nets may also reduce bycatch rates of this species. While Orphanides (2009) did not find that rigged net height was retained as a significant explanatory variable of harbour porpoise bycatch in sink gillnets in the USA Atlantic, the dataset analysed in the study combined both

gillnets and tangle nets. Results of analysis of bycatch by metier in UK static net fisheries show that relatively high harbour porpoise bycatch rates can be observed in both gillnets and tangle nets. However, these gear types vary in a number of characteristics other than rigged height, the most obvious of these being mesh size. Therefore if statistical models are constructed for combined data from both these two gear types, it is likely that there will be a stronger signal from other gear characteristics than rigged height.

There is some uncertainty around the values of rigged net heights recorded for UK static net fisheries, as onboard observers do not have the opportunity to directly measure nets and often rely on information from the skipper about the rigged net height. While the average height of hake nets reported by Tregenza *et al*. (1997a) was similar to those recorded by observers in UK dataset used for modelling, Vinther (1999) reported that Danish hake nets could reach heights of up to 9m. The actual fishing profile of static nets, relative to their rigged height will be affected by a number of factors including the amount of flotation used, the solidity of the net material, the twine diameter and the current speeds they are subjected to (Stewart & Ferro 1985). Therefore, while the covariate rigged net height was retained for the best model describing harbour porpoise bycatch rates in ICES Area VII, this covariate is unable to reflect the actual fishing profile of the nets observed in these fisheries. Indeed little information exists on the underwater fishing behaviour of static nets.

2.5.4   Mesh size

Mesh size was found to have significant positive relationship with harbour porpoise bycatch rates in gillnets in ICES Area VII. While a positive trend between mesh size and harbour porpoise rates in ICES Area IV was also observed this relationship was not significant. In contrast mesh size was not retained as an explanatory variable of harbour porpoise bycatch in tangle nets in either ICES Area IV or VII. For tangle nets in ICES Area IV, model results were driven by experimental trials in ICES subdivision IVb where the majority of hauls were of nets with a mesh size of 267mm. Although mesh size was not retained in the best model for tangle nets in ICES Area

VII this model was unstable and it is not clear whether the retained variable floats present is actually a proxy for mesh size.

Mesh size has previously been shown to have a positive relationship with bycatch rates of harbour porpoises (Palka 2000; Orphanides 2010), bottlenose dolphins (Palka and Rossman 2001) and loggerhead turtles (Murray 2009) in static net fisheries. Palka & Rossman (2001) estimated highest bycatch rates of bottlenose dolphins caught in USA Mid-Atlantic sink gillnet fisheries, in mesh sizes greater than 155mm, and intermediate bycatch rates for mesh sizes of 127-155. Murray (2009) reported a positive relationship between loggerhead turtle bycatch and mesh size in the same fishery, with 20% of the variation in loggerhead turtle bycatch rates being explained by mesh size.

As previously stated, there were too many missing values of twine diameter to include this gear characteristic as a covariate in models of cetacean bycatch in UK static net fisheries. In general as mesh size increases there is a concurrent increase in twine diameter. However, depending on the target fish species, nets of similar mesh size can have different twine diameters, for example sole and hake nets in the UK. While thinner twine diameters have been shown to increase capture rates of some target fish species (He 2006), the thinner the twine the more susceptible the net is to damage, thereby reducing its fishing life. Also, thinner twines have a larger size selection range as the material is more easily elongated by a fish trying to push through it (He 2006).

Two separate experimental trials in the UK suggest that twine diameter may influence harbour porpoise bycatch rates (Northridge et al. 2003). The first trial compared bycatch rates in two nets, one with a twine diameter of 0.4mm the second with a twine diameter of 0.6mm. Bycatch rates were significantly lower in the thin twine nets, and the number of holes recorded by observers was also higher for this net type. However, these results are confounded by the thinner twine net also having a smaller mesh size. A second trial compared two nets with twine diameters of 0.6mm and 0.67mm, and recorded bycatch rates were higher in the thicker twinned net. However this net also had a smaller mesh size compared to the control net. Palka (2000) found that that highest bycatch rates in the Mid-Atlantic sink gillnet fishery were observed in nets with twine diameters of 0.57 and 0.62 mm.

While no harbour porpoise bycatch was recorded in 508 observed hauls in sole nets in ICES Area VII, this métier had a similar average mesh size to nets targeting hake (124mm and 123mm) respectively. Likewise, no harbour porpoise bycatch was recorded in the UK sole nets in ICES Area IV, although only 51 hauls were observed, so the sample size is too low to conclude anything. Although observer coverage in the Danish sole fishery was also low, Vinther (1999) concluded that the lack of harbour porpoise bycatch in that fishery could be a result of the small mesh size used, or a result of the lower profile of these nets or the "less robust netting material used". As previously stated, while mesh sizes of hake and sole nets are similar, the latter metier has a much lower profile and thinner netting material. However, five harbour porpoise have been reported bycaught in UK sole nets rigged as trammel nets. While the height of these sole nets is similar to those consisting of a single wall of webbing, the outer mesh sizes are much larger. Therefore, the probability of harbour porpoise entanglement in trammel nets fishing for sole is likely increased by this large mesh size. The lack of observed harbour porpoise bycatches in gillnets targeting sole in both UK and Danish fisheries suggests that some characteristic of these nets may result in a reduction in the probability of bycatch. Whether this is due to the low profile of these nets, or the thin twine diameter of the meshes (and therefore lower breaking strain) remains unclear.

2.5.5   Common dolphin bycatch in static nets.

The best model predicting common dolphin bycatch in static nets in ICES Area VII retained the variables soak time, mesh size ($\geq$ 203mm) and month (October-December), and an interaction between mesh size and soak time. Examination of the data from this model showed significant autocorrelation between hauls which was driven by the extremely clumped nature of observed bycatches of common dolphins. Almost half of all common dolphin bycatches were recorded from four successive trips on the same vessel over a period of 8 days. Nine of the common dolphins were bycaught in tangle nets (eight in one trip) while two were caught in wreck nets. It is the nine animals caught in tangle nets by this boat that drive the apparent relationship between common dolphin bycatch and mesh sizes greater than 203mm. However, it is clear that common dolphins were susceptible to being caught in both of the two gears this boat fished over a small spatial and temporal scale. The two types of nets

deployed are very different in their gear characteristics. The wreck nets are 7.8m tall, with a mesh size of 130mm and one was 376m in length while the other was 752m in length. The tangle nets used by this boat had a mesh size of 300mm, a fleet length of 1152m and were 1.5m tall. Over half of the bycatch events occurred between October and December. An increase in sightings rates of common dolphins during winter months in the Western approaches of the English Channel has been reported as well as concurrent increase in strandings of this species. Tregenza et al. (1997a) found a peak in sightings rates between November and December when investigating the bycatch of this species in the UK and Irish hake gillnet fishery. Four common dolphins were observed bycaught during the Tregenza et al. study, of which one was alive and fell out of the net as it was being hauled. Given that this animal was still alive during haul back, and common dolphins had been observed in the vicinity of nets as they were being shot in two of the three times bycatch had been recorded, the authors suggested three possible mechanisms for common dolphin bycatch in these nets. 1). That animals become entangled during hauling or shooting of nets. 2) That the risk of entanglement is increased if dolphins are engaged in "playing" with nets. 3). That the observed responsive reaction of attraction to boats may increase the probability of coming into contact with nets. The observation of common dolphin bycatch in two very different gears deployed by the same boat in the same area does suggest that it is the animals being there and maybe interacting with gear during shooting or hauling that effects the bycatch and not the gear characteristics. The 20 common dolphin bycatch events observed in UK fisheries occured in nets with soak durations ranging from 12 –240 hours, and over half the bycatch *events* were in nets soaked for 24 hours or less. Therefore the probability of entanglement for this species does not seem to be related to the length of time that gillnets or tangle nets are deployed.

2.5.6   Conclusions

The results of statistical analysis of independent onboard observer data in UK static net fisheries found relationships between harbour porpoise bycatch rates and a number of covariates describing gear characteristics. These were fleet length, net height and mesh size.

In the US Mid-Atlantic sink gillnet fisheries, regulations to limit harbour porpoise bycatch are based around seasonal area closures or gear modifications for specified fisheries (Orphanides 2009), including a maximum fleet length allowed for different size mesh nets. For gillnets with a mesh size of 180-460mm, these regulations specify a restriction of net lengths to 1.46km, a limit on the number of nets per vessel and a minimum twine diameter of 0.9mm. Gear modifications include the mandatory use of tie downs, with a maximum length of 1.2m and spaced at 7.3m intervals. For gillnets with a mesh size of 130mm-180mm, net lengths are restricted to 0.914km the twine diameter must be at least 0.81mm and the use of tie downs is prohibited. In the UK, high bycatch rates of harbour porpoise in short wreck nets targeting cod were recorded in both ICES Area IV and VII and in longer nets targeting hake in ICES Area VII. However, an increase in the length of hake nets from an average of 1.6km (1992-1996) to an average of 4.8km (recorded since 2004) has not resulted in an increased bycatch rate in this fishery. It is not clear from these results how limiting fleet length, especially in wreck net fisheries which already use short fleet lengths, could be used as a management measure to reduce harbour porpoise bycatch in UK static net fisheries.

The relationship between harbour porpoise bycatch and rigged net height in ICES Area VII, and the lack of observed bycatches in lower profile sole nets, suggests that this could be an interesting gear characteristic to explore for its potential at reducing small cetacean bycatch. While the use of tie downs, which lower the fishing profile of gillnets, have been associated with a reduction in harbour porpoise (Palka 2000) and common dolphin bycatch (ICES 2011), they have also resulted in higher bycatch rates of Atlantic sturgeon (ICES 2011) and sea turtles (Price & Van Salisbury 2007). However, an elimination of tie downs and the use of lower profile nets has been shown to reduce sea turtle bycatch whist maintaining acceptable levels of target catch (Price & Van Salisbury 2007). It is unclear whether the observed reduction of cetacean bycatch in nets with tie downs is a result of the lower profile of these nets, or the increased bagginess of the netting material. As tie downs are associated with increased bycatch rates of other protected species the effects of low profile nets on cetacean bycatch rates should be tested, as this gear modification has the potential to work for a number of species.

Low profile nets (relative to standard nets) have also been successful at reducing the bycatch rates of Atlantic cod in multi-species gillnets in the Gulf of Maine, whilst increasing catches of flounder (He 2006a). However, catch rates of white hake were also substantially reduced in the low profile nets. The standard net used in this flounder fishery was 4.5m high, which is similar to net heights recorded in wreck nets and cod and hake nets in ICES Area VII. Whilst the reduction of heights in UK cod and wreck nets could be tested as a means of mitigating harbour porpoise bycatch, the results of He (2006a) indicate that the effect of net height on target catch rates may not make this an economically acceptable mitigation strategy for these fisheries. In contrast, tangle nets generally fish close to the sea floor and may lie down on the substrate for some periods when they are deployed. Given the behaviour of the demersal fish and crustacean species targeted by these nets in UK waters, it is possible that a reduction in net height would not substantially impact commercial catches.

Mesh size was found to have a positive significant relationship with harbour propoise bycatch in gillnets in ICES Area VI and VII. While there also appeared to be a positive relationship between common dolphin bycatch and mesh sizes in static nets in ICES Area VII, it was clear that eight animals caught in the same trip in 2005 drove this result. A positive relationship between mesh size and bycatch rates has previously been shown for harbour porpoise (Palka 2000, Orphanides 2010), bottlenose dolphins (Palka and Rossman 2001) and loggerhead turtles (Murray 2009). The mesh size used in static net fisheries determines the size and type of targeted species caught (He 2006b). The twine diameter used in gillnets is often related to the mesh size. It is possible that reducing the twine diameter of meshes, which would lead to a concurrent reduction in the breaking strain of the net, may reduce bycatch by increasing the probability that a small cetacean could break free of a net after entanglement had occurred. However, the practicalities of using thinner twined gillnets will depend on the operational characteristics of the fishery. If nets are shot over rough ground then decreased twine diameter will reduce the durability of these nets when they become snagged and would likely reduce the fishing life of these nets. In addition, reducing the breaking strength of twines may increase the likelihood of netting becoming lost at sea, therefore resulting in other ecological impacts that are related to ghost fishing,

Finally gillnets can vary in the hanging ratio used when the net material is rigged to the float line and lead line. The hanging ratio will determine how solid or loose the net material will hang. Larger hanging ratios will result in the net hanging in folds and are commonly used for nets which aim to entangle target species. Data on the hanging ratio of nets were not available for all observed hauls and was therefore not considered as an explanatory variable in analysis of the UK observer data. However, a recent study in the US has shown no significant difference in cetacean bycatch rates with different hanging ratios (AIS Inc. 2010).

Explanatory variables relating to environmental factors have been retained in a number of studies that have used statistical models to investigate catch rates of non-target species. Depth has been retained as an explanatory variable of harbour porpoise bycatch rates in US North-eastern and Mid-Atlantic States sink gillnet fisheries (Orphanides 2009, Palka 2000). Orphanides (2009) suggested that the significance of this variable, as well as season, reflected a north-south seasonal movement of harbour porpoises interacting with gillnets between the 55 and 110m depth contours, and cited Read and Westage (1997) who reported that satellite data collected from free ranging harbour porpoises showed animals were most commonly distributed around the 92m depth contour. In the UK data, the retention of depth in the two best models for harbour porpoise bycatch in gillnets in ICES Area IV and VII is a reflection of the fishing areas used by wreck netters and the UK hake fishery.

Season has also been found to be associated with harbour porpoise bycatch rates in a number of studies (Orphanides 2009, Palka 2000, Vinther 1999). While season was not retained in any of the models from UK observer data, highest harbour porpoise bycatch rates in ICES Area IV were observed in nets targeting cod between July and September. Vinther (1999) also reported highest bycatch rates in these months (when fleet length was used as a measure of fishing effort), and proposed that the observed reduction in the length of bycaught harbour porpoise during this period may indicate that calves are being caught. Seasonal movements of harbour porpoise have been reported between the inner Danish Sea and the North Sea (Teilmann et al. 2004) and in the German Baltic (Verfuß et al. 2007). Highest bycatch rates of common dolphins in ICES Area VII were recorded during winter months which is when an increased abundance of this species in the Western English Channel is reported. Palka (2000)

also reported a relationship between harbour porpoise bycatch and sea surface temperature, with decreased bycatch rates in years with warmer water temperatures.

While environmental, spatial and temporal covariates are often retained when modelling the bycatch of different species, it can be difficult to judge whether these variables are related to changes in distribution and abundance of the bycaught species, or may be retained as a proxy for some behaviour or characteristic of a fishery that is not captured in the data. While Orphanides (2009) concluded that the spatial and temporal distribution of harbour porpoise had the biggest effect on bycatch rates, the main aim of this model was to produce bycatch estimates for unobserved fleet segments. These estimates require that models contain covariates that can be extrapolated to the entire fleet and therefore data may be stratified in the most appropriate manner to achieve this. Such stratification can result in possibly important gear characteristics not being retained in model selection. When data for gillnets and tangle nets were grouped, the models from ICES Areas IV and VII did not retain gear related covariates at a level that could be used to identify specific gear characteristics that could potentially be modified to reduce bycatch. However, by sub setting data from these two areas by net type, the total number of observed bycatch events was reduced for each model, thereby reducing the number of covariates that could be retained without over-paramaterising the model, or causing model instability.

While analysis of observer data can provide insights into gear characteristics which may be modified to reduce bycatch of non-target species (Palka 2000), and to assess the effects of such modifications (Gilman et al 2008), it is clear from the analysis of UK observer data that the ability of models to find such relationships is confounded by relative rarity of bycatch events and the distribution of observer effort in fisheries where no bycatches are recorded. In order for explanatory models to have enough power to identify specific gear characteristics or fishing practices related to higher bycatch rate a sufficient number of bycatch events in individual fisheries are required. Over-parameterisation of models will occur if the number of observed events per number of parameters is low (Peduzzi et al. 1996).

From the analysis of UK observer data and other studies, it is apparent that the metric of fishing effort used (e.g. haul, km net hours or amount of landed fish) when

modelling bycatch may affect the relative significance of different explanatory variables. For example while harbour porpoise bycatch rates per haul was the same for wreck nets in ICES Area IV and VII (0.01 animals per haul), bycatch rates per 10 km net hour were much higher in wreck nets in ICES Area IV (0.02 v. 0.007). Likewise, when this metric was used, bycatch rates in gillnets targeting cod in ICES were higher than bycatch rates in gillnets targeting hake (0.007 v 0.004). In contrast, if the metric used was observed number of bycatches per haul then bycatch rates in the hake fishery were higher (0.04 v 0.01). In the USA, total fish landings have been found to produce unbiased estimates of harbour porpoise bycatch rates in sink net fisheries, when compared to measures of effort such fleet length and soak time (Rossman & Orphanides 2009). However, Vinther (1999) reported that while harbour porpoise bycatch rates were associated with higher cod CPUE in Danish wreck net fisheries, in other Danish cod fisheries, highest harbour porpoise bycatch rates were observed in nets with low cod CPUE. A clear linear relationship was not found between harbour porpoise bycatch rates and fish catches in UK gillnet or tangle net fisheries, but data on fish catches were limited. Given the results reported by Vinther (1999) it is clear that relationship between bycatch rates and catch rates of target species will be related to the specific gear type used. Therefore, if cod landings are used to estimate harbour porpoise bycatch, and it is not possible to stratify landings by short or long cod nets, then estimated bycatch rates will be biased. If this metric was used as an offset in statistical models to identify factors influencing bycatch rates in UK set net fisheries model results may have been different to those obtained using km net hours as a measure of fishing effort. As data on target catch rates were not available for all observed hauls, the effect of using this metric as on offset on model results could not be tested.

Finally, the unit of observation used in this analysis was a haul. While dispersion parameters estimated by models with quasi-Poisson error distributions did not indicate over dispersion in the data, due to the low counts of harbour porpoises bycaught per haul, any over dispersion in the data may have been underestimated. Results of autocorrelation of model residuals showed that bycatches were clustered within trips, meaning that using a haul breaks the Poisson model assumption of independence. However, if data were modelled at a trip level it would not be possible to include many of the gear characteristics we wish to investigate, as these would have to be

averaged for all hauls across the trip. Autocorrelation can result in more importance being given to similar characteristics recorded in the same trip, and in effect means that there is less data available for analysis.

Results of analysis of gear characteristics effecting the bycatch rates of harbour porpoises and common dolphins in UK gillnet fisheries indicate two gear characteristics which could be investigated for bycatch mitigation. These are twine diameter and net height. However, the actual height of bottom set gillnets is hard to measure and a better understanding of how different gear characteristics affect the fishing profile of nets is needed. These issues are addressed in Chapter 3.

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3. Chapter 3: An investigation of the underwater fishing behaviour of gillnets

3.1 Abstract

Changes to fishing gear or practices may present the best option for minimizing the bycatch of non-target species. While a number of gear modifications have been tested to reduce bycatch rates of large marine vertebrates in bottom set gillnets, the effects of such modification on the underwater fishing behaviour of experimental nets have rarely been considered. Depth loggers were used to record the underwater fishing behaviour of five gillnets with different gear characteristics. Results showed that the proportion of net area fished, relative to the theoretical rigged net area, ranged from 0.36 to 0.65 depending on the specific gear characteristics of the net. The effect of how gear modifications may change the fishing heights of experimental nets have rarely if ever been considered for paired bycatch mitigation trials. Results of simulated experiments showed that if a gear modification leads to a reduction in the fishing profile of an experimental net, and this reduction is not accounted for, then the significance of observed differences in bycatch rates between control and experimental nets may be overstated and the power to detect a pre-specified reduction in bycatch rates will be decreased.

3.2 Introduction

Gillnets are one of the most important fishing gears used worldwide (He 2006b), accounting for the third largest global quantity of fish catch after seine and trawl nets (Watson et al. 2004). However, the study by Watson et al. (2004) likely underestimates the importance of gillnets as smaller vessels, which catch less fish individually, but are far more numerous, commonly use this gear. These gears comprise of at least a single wall of mesh, with a buoyant float line (also termed headline) and a weighted lead line that can either be anchored at the surface or to the sea bottom (gillnets) or drift at the surface or above the sea bottom (drift nets). These nets are designed to be as invisible as possible so that fish are unable to detect them

and will swim through them and become captured (He 2006b). The mesh size used in sink gillnets will affect the mechanism by which target catch is retained. Smaller mesh nets typically enmesh fish or trap them behind the gill rakers (gillnet) whilst larger mesh nets generally entangle target species (tangle nets). The hanging ratio of the net, which effects how slackly the netting material hangs, also defines the capture mechanism. For all nets, the modal length of fish caught increases with mesh size, and both gillnets and tangle nets are considered to be highly selective for specific length ranges of target species (He 2006b). However, large marine vertebrates are also frequently caught in this type of fishing gear.

The majority of global marine mammal bycatch is believed to occur in gillnets (Read et al. 2006) and there is growing evidence that the bycatch rates of marine turtles in this gear type may have important impacts on the conservation of these species (Peckham et al. 2007, Zydelis et al. 2009, Gilman et al. 2010). Likewise, a number of sink gillnet fisheries in the USA have been observed to catch large numbers of seabirds (Moore et al. 2009) and shark species (Thorpe and Frierson 2009) and it is fair to assume that such bycatches occur in gillnet fisheries globally. In response to these high bycatch rates, a number of gear modifications have been developed and tested to reduce accidental capture and mortality of these taxa in bottom set gillnets. Gear modifications tested include making nets more visible (Melvin et al. 1999), reducing the fishing profile either by reducing the rigged height (Price and Van Salisbury 2007) or through the use of tie downs (Palka 2000), changing the hanging ratio of the net (Anon. 2010) and, specifically for cetaceans, making the nets more acoustically refelctive to echolocation clicks (Trippel et al. 2003, Larsen et al. 2007).

Results of an analysis of factors effecting harbour porpoise bycatch in bottom set gillnets in the UK (Chapter 2) showed that net height (calculated as mesh size multiplied by number of meshes) was a significant variable for predicting bycatch. While this result suggests a possible avenue to explore to reduce harbour porpoise bycatch rates, the covariate net height which was retained by the model only reflected the recorded rigged height of gillnets. Therefore, this measure does not take into account other gear characteristics or environmental factors which have been shown to affect the actual net profile of bottom set gillnets as they fish. Stewart & Ferro (1984) investigated the affects of water speed and gear characteristics on the fishing profile

(measured as the height of the float line of the net) of short gillnets (~5m) in a flume tank. Gear characteristics tested included mesh size, net length, net height and amount of float line buoyancy. They found that twine area and current speed had the greatest effect on netting drag and therefore float line height. In addition, the stiffest monofilament net tested had the highest recorded value of drag to twine area and the effect of current speed on float line height was greater when nets were set perpendicular to water flow. Stewart (1998) conducted a field trial to compare the results of these experiments with data collected from a fleet of nets fished either as two fleets of 110m in length or one fleet of 220m. The difference in pressure recorded by manometers, deployed on the float line and lead line, was used to calculate the float line height of the fleet of nets, set either parallel or perpendicular to the surface tidal current during various periods of the tidal cycle. The mean float line height for the shorter fleet of nets was 65% of rigged height when the nets were shot parallel to the water flow, and 50% of rigged height when shot perpendicular to the tidal flow. For the longer fleet of nets, mean float line heights were 45% of rigged height when set parallel to the flow and between 35% and 45% of rigged height, dependent on the position of the manometers when set perpendicular to the flow. Additional work by Matuda (1998) produced a formula for predicting the float line height of bottom set gillnets shot across the water flow and found close agreement with the results of Stewart and Ferro's flume tank measurements. The two dimensional models developed by Stewart (1998) and Matuda (1998) have been improved by taking into account three dimensional drag forces exerted on bottom set gillnets. Shimizu et al. (2007) used a net-shape and loading analysis system (NaNL) to predict the dynamic behaviour of a bottom set gill net and then used depth loggers to collect *in situ* data on float line heights. They also found that as current speed increased net height was reduced and the lowest float line height was approximately half the initial height in water and that current speed had a greater effect on net height when perpendicular to the net. These studies show that the true fishing profile of nets relative to their rigged height are a function of current speed and direction and the total drag of the net, and that net drag is a function of mesh size, twine diameter, buoyancy, net length and bridle length.

Direct investigation of the effects of reducing the profile of bottom set gillnets on the bycatch rates of non-target species have only been conducted in a number of trials. He

(2006a) found that reducing the profile of gillnets resulted in higher catch rates of targeted flatfish species, whilst achieving a desired reduction in the catch rates of Atlantic cod (*Gadus morhua*). In a trial in Australia, a reduction in net profile resulted in a significant decrease in the bycatch of a number of fish species but did not reduce the catch of the target species dusky flathead (*Platycephalus fuscus*) (Gray et al. 2005). Bycatch rates of marine turtles have also been reduced in low profile nets (Price and Van Salisbury 2007), though a reduction in target catch was also reported. The use of tie downs, which effectively reduce the fishing profile of gillnets has also been associated with a reduction in the bycatch rate of harbour porpoises (Palka 2000), and common dolphins (ICES 2011).

The effect of gear modifications on bycatch rates are usually tested by comparing paired nets where standard and modified fishing gears are fished in the same area under the same operational and environmental conditions. The aim of such experiments is to try to keep as many variables as possible constant so that any changes in bycatch rates in experimental gear can be directly related to the specific gear modification. Sound experimental design would include a power analysis to determine, given a known background bycatch rate, how many hauls of the control and experimental net would need to be observed to detect a specified difference in bycatch rates with sufficient statistical power.

The statistical power of a test is 1- β, where β is the probability of getting a Type II error. A Type II error occurs if the null hypothesis (that control and experimental fishing gear catch the same number of animals) is rejected, although it is true. While it is unclear whether the probability of cetacean bycatch rates are directly related to the fishing profile of static gillnets, changing just one gear characteristic such as the amount of flotation used in an experimental net is likely to change the height at which the float line of the net will fish (Stewart and Ferro 1985). If gear modifications result in large changes in the profiles of experimental nets, then the results of experimental bycatch mitigation trials testing single gear characteristics may be compounded if the probability of entanglement is not equal for modified and standard gillnets. Such an effect, if not accounted for, will increase the probability of a Type II error occurring and lead to incorrect conclusions about the statistical significance of results.

A number of bycatch mitigation trials have been conducted in the UK to investigate the effect of different gear characteristics on harbour porpoise bycatch rates (Northridge et al. 2003). These have looked at the effects of twine diameter, acoustically reflective nets and tensioned nets on bycatch rates. The first aim of this chapter is to utilise self-contained depth loggers to measure the underwater fishing behaviour of the modified and standard gillnets originally used in these experimental trials and compare the proportion of net area, relative to rigged area, that each net fishes. The second aim is to assess how differences in net profile, if unaccounted for may effect the interpretation of results from paired experimental trials and to investigate how a change in fishing profile affects calculations of power analyses, based on an assumption that there is a linear relationship between cetacean bycatch and the fishing profile of static gillnets.

3.3 Methods

3.3.1 Measurement of float line heights

A pair of Star ODDi DST-milli© depth temperature loggers (www.star-oddi.com) were attached to the mid-point of each gillnet during fishing trials. These loggers can be programmed to record temperature and depth at desired intervals, and are small (38mm x 13mm) and light (5g in water); therefore their effect on the behaviour of the float lines of gillnets should be negligible. Purpose built lightweight steel housings were produced to protect the tags during fishing operations (Fig 1a&b).
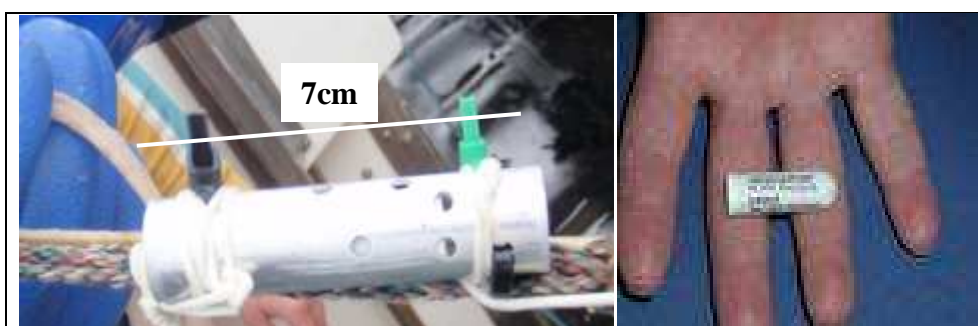
Figure 1 a & b: purpose built housing for tags with tag inside, and size of tag relative to hand.

Tags were attached at approximately the midpoint of the float line and lead line of each net, or the bridle between two nets using cable ties (Fig 2). Each logger was set to record pressure data (in bar) at ten minute intervals and temperature at 61 minute intervals. The loggers record depth measurements with an accuracy equivalent to +/- 0.4m.

Figure 2: Schematic of depth logger attachment on a gillnet. Not to scale.
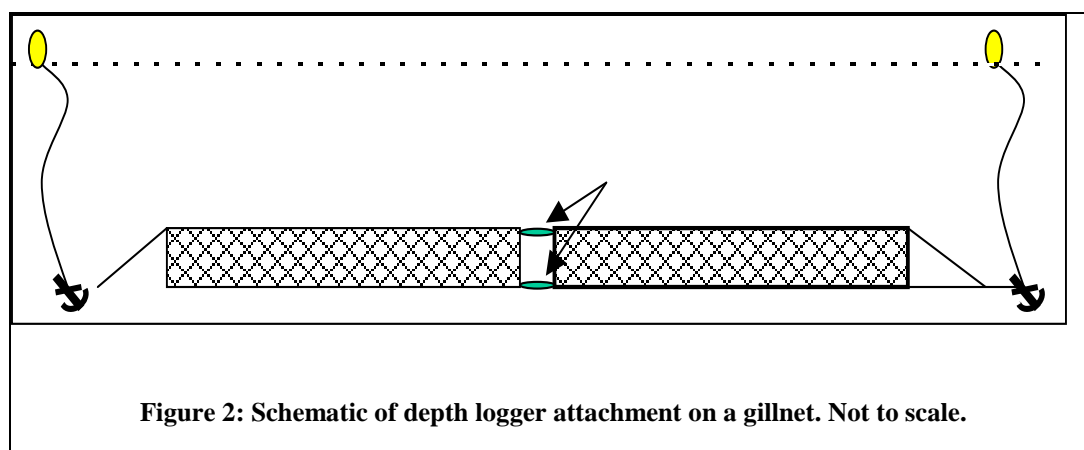
By placing a logger on the lead line, the effect of tidal height on the pressure recordings from the float line could be ignored, as the active fishing height of the net can simply be calculated by subtracting measurements recorded by the float line from those from the lead line. Data were downloaded from the tags using the Star Oddi SeaStar© program and summary statistics were calculated before exporting the data for further analysis. After inspection of the data, all depth data recorded during periods when nets were being hauled, shot or on deck, were removed before further analysis. Tidal heights for the period of the trial were predicted using the tidal software POLPRED (www.pol.ac.uk).

3.3.2 First field trial Bridlington, North Yorkshire.

Three fleets of gillnets were deployed between the 19th of July and 11th of August 2009 in Bridlington Bay, North Yorkshire (54° 00 N 0° 04W) during an experiment investigating the echolocation behaviour and occurrence of harbour porpoise in the vicinity of gillnets. Methodology and results for this experiment are reported in more detail in Chapter 4. Each net was identical in length (200m), mesh size (266mm) and

net height (7.5 meshes ~ 2m rigged height) but were rigged with different amounts of flotation. Tier 1 (Single) was rigged as a standard skate net with a 10mm float line and 4.5mm lead line. Tier 2 (Double) was rigged with a 12mm float line and 9mm lead line. For Tier 3 (Cigar), the standard float line was replaced with a 10mm braided polypropylene float line, with 6" polystyrene cigar floats attached at 5 metre intervals. Table 1 provides a summary of gear characteristics of each of the nets.

Tier Code	Mesh size	Twine diam (mm).	Height (in meshes)	Float line diameter	Lead line diameter
1 - Single	267mm	0.6	7 ½	10 mm	5 mm
2 - Double	267mm	0.6	7 ½	12 mm	9.4 mm
3 - Cigar	267mm	0.6	7 ½	9.5 mm	3.6 mm

Table 1: Summary of gear characteristics of gillnets deployed off Bridlington, North Yorkshire.

All nets were shot parallel to the current in depths of approximately 14 metres over a sandy bottom. Figure 3 shows the location of net deployments.
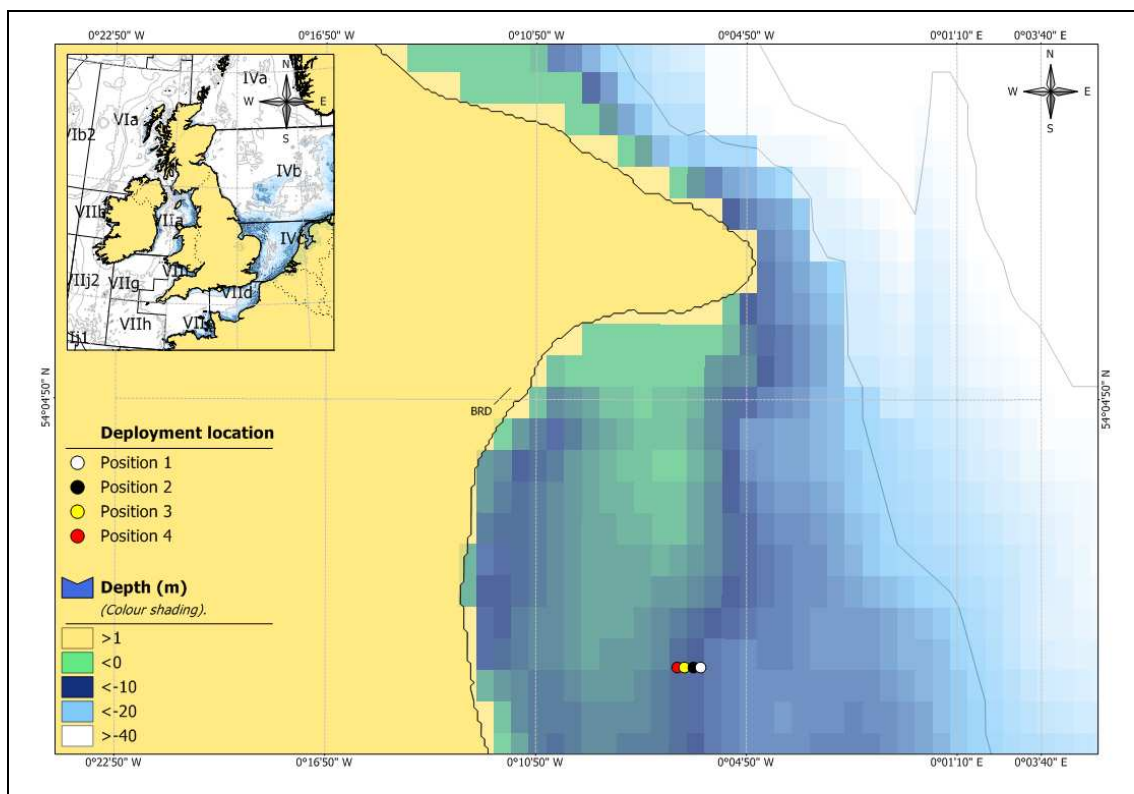
Figure 3: Locations of four positions where nets were deployed in Bridlington Bay, North Yorkshire.

The location of the nets was rotated three times during the experimental trial. Specific details are given in Chapter 4.

3.3.3 Second field trial, St Andrews Scotland

A further trial was conducted between the 18th and 21st of September and the 24th and 27th of September, 2009, in St Andrews Bay, Scotland (56° 21' N 2° 45' W). Three nets were deployed in each trial; a standard skate net, a standard bass net and a gillnet modified with barium sulphate ($BaSO_4$) (Fig 4)
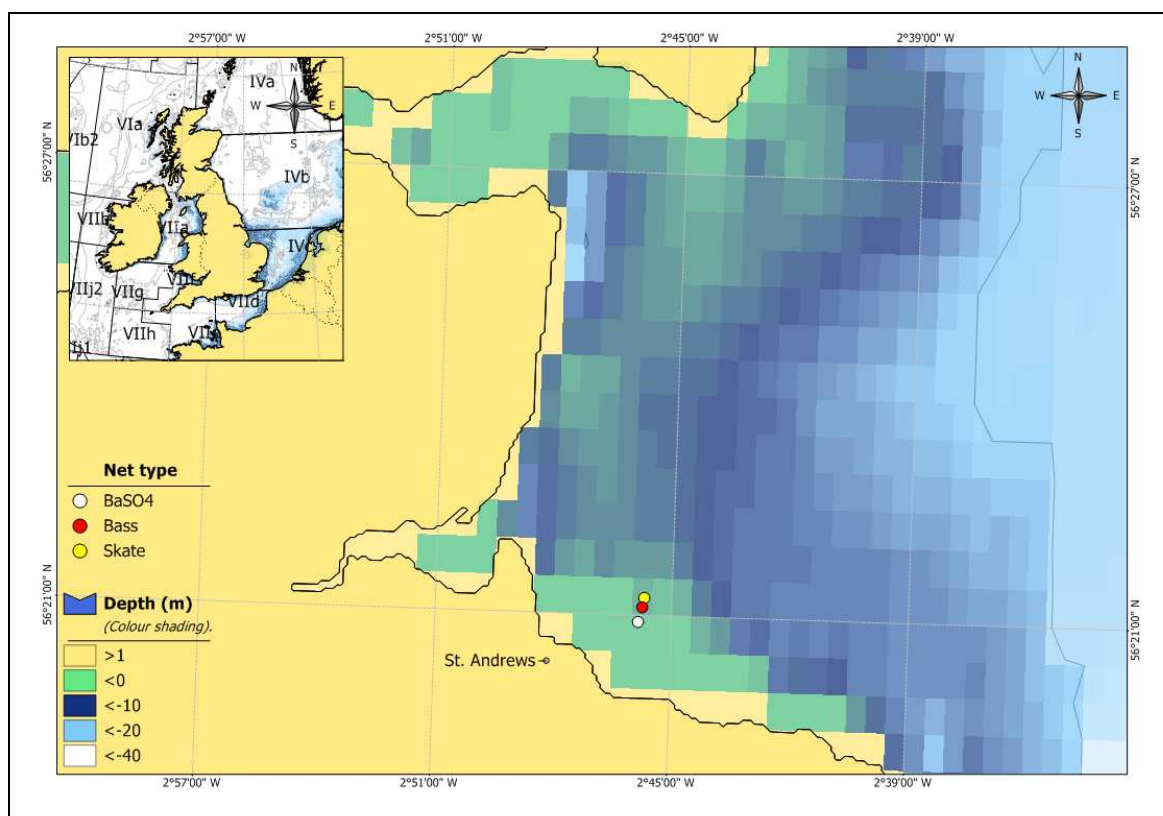
Figure 4: Location of nets deployed in St Andrews Bay, Scotland.

Barium sulphate is an inorganic compound and fine particles of this compound have been mixed with nylon to produce a modified gillnet with the assumption that the addition of this material would make gillnet meshes more acoustically reflective to cetacean echolocation clicks (Trippel et al. 2003). Nets were deployed by FV Rose III and were shot and hauled by hand. Each type of net had previously been used in experiments investigating the effect of different gear characteristics on harbour porpoise bycatch rates (Northridge et al. 2003). Tier 1 was a standard skate net of length 100m, a rigged height of 2m and a mesh size of 267mm. Tier 2 was a standard bass net with length 90m, rigged height of 4.5m and a mesh size of 90mm. Tier 3 was a gillnet modified with $BaSO_4$ which had a length of 108m, a rigged height of 3.5m

and a mesh size of 241mm. The rigged height of the net is calculated by multiplying the number of meshes the net is high by the stretched mesh size. The rigged height of each net was the same as had been used in the experimental trials conducted by Northridge et al. (2003). Table 2 summarises the gear characteristics of the three nets.

Tier Code	Mesh size	Twine diam.	Height (in meshes)	Float line diameter	Lead line diameter
1 - Skate	267 mm	0.6	7 ½	10 mm	5 mm
2 - Bass	90 mm	0.4	50	2 x 6mm	4 mm
3 – BaSO$_4$	241 mm	0.67	14 ½	9 mm	4 mm

Table 2: Summary of gear characteristics of gillnets deployed in St Andrews Bay, Scotland.

In addition, a single Aquamark™ pinger was deployed to minimize the likelihood of cetacean bycatch. As each tier consisted of only one net panel, data loggers were deployed at the midpoint of each net. Each net was shot parallel to the current in a depth of approximately 17 metres on a sandy bottom. Figure 4 shows the location of all net deployments.

3.3.4 Data cleaning and preparation

Data were downloaded using Seastar software and exported to Microsoft Excel. Any measurements taken whilst the nets were out of the water, either prior to shooting or on deck during hauling were removed. Net height was then calculated as the difference in depth recorded by each pair of tags deployed on a single net. Tidal data including tide height, current speed and current direction were predicted for periods when nets were deployed using POLRED Offshore computation software.

3.3.5 Analysis

Because measurements collected by the depth loggers at 10 minute intervals are serial data, they will be auto correlated. If this autocorrelation is not accounted for, analysis of these data could result in estimates with smaller confidence intervals than would be expected in the data were independent. To account for this autocorrelation, data were grouped at the most appropriate temporal level (e.g. hourly, six hourly) for analysis.

3.3.5.1   Comparing the active fishing height of float lines

To investigate whether the recorded float line heights of deployed gillnets were significantly different, a generalized linear model (GLM) was constructed using data collected by loggers at 6 hour intervals for the duration of each deployment. Data on the difference of each headline measurement to the rigged net height were used so that no non-negative values were added to the data. The response variable was the difference in measured float line height from the theoretical rigged height. The error structure of the response variable was assumed to be Gamma distributed and was modeled through an inverse link function. The explanatory variables net type, current speed and current direction were offered to the model using forward and backward selection, governed by AIC. Current speed and current direction were predicted using the POLPRED Offshore computation software as empirical data on current speed were not collected during the field trials.

Bootstrapped Kolmogorov-Smirnov tests (R package matchings) were used to investigate whether there was any difference in the distribution of float line heights recorded for individual nets between different deployments. The dataset tested contained float line height measurements recorded at hourly intervals. This methodology was also used to determine if the distribution of differences between pairs of nets were similar for all deployments (e.g. Double v Single, Double v Cigar, Single v Cigar).

The mean float line heights of each net during each deployment were calculated using data collected at ten minute intervals. These data were also used to calculate the proportion of the theoretical net area fished by each net during each deployment. The two dimensional theoretical fishing area of a gillnet can be calculated as the length of the net multiplied by the rigged height of the net. While mean values of recorded float line heights provide an overview of the average fishing profile of a net, they are not informative for ascertaining how the profile of the net changes relative to the theoretical rigged height as the net is fishing. Therefore the two dimensional area that each net fished relative to the theoretical rigged height was calculated for each 10 minute measurement of float line height.

3.3.5.2 Analysis of the effect of reduced net profile on statistical significance and power

The relationship between the actual fishing profile of a gillnet and cetacean bycatch is unclear. However, if a reduction in net profile leads to a reduction in bycatch rates then the statistical significance of observed bycatches may be overstated if a gear modification indirectly leads to a reduction in the fishing profile of an experimental net. In addition, if such effects are not considered during experimental design then a greater number of hauls than assumed may need to be observed to have sufficient power to detect a specified reduction in bycatch rates in experimental fishing gear.

Confidence intervals around the underlying observed bycatch rates of porpoises in an experimental net relative to a control net can be used to determine which combinations of counts of bycatches in each net are significantly different. A simulation was run of experimental trials comprising of 200 hauls of both control and experimental nets, with a background bycatch rate of 0.04 animals per haul. A data set was then constructed that contained the outcomes of all combinations of each net catching between zero and eight animals per trial, resulting in a total of 81 trials. Confidence intervals were obtained using a modified version of the R function riskcoreci (Mike Lonergan, SMRU). This function calculates confidence intervals around the ratio of bycatch rates between control and experimental nets given the number of animals caught in each treatment per simulated experiment. The coverage of the confidence interval is 0.9 giving a 5% chance of a Type I error in the 1-tailed test. If the upper confidence level is greater than or equal to one the null hypothesis is rejected. If the experimental net was assumed to have a 25% reduction in fishing profile relative to the control net then any confidence interval with an upper bound of less than 0.75 indicates that a ratio of capture rates in both net types remains significant, even if 25% of the difference in bycatch rates is a result of the difference in net profiles between the two treatments.

The upper bound of the confidence interval of each simulated experiment was then used to determine which combinations of counts of animals in the control and experimental net were significantly different given equal and reduced float line heights in the experimental net relative to the control net (Table 3).

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Reduction in float line height - experimental net relative to control net.	Cut off point for significant difference ~ upper limit of confidence interval
0%	< 1
25%	< 0.75
40%	< 0.6
50%	< 0.5
60%	< 0.4

Table 3: Cut off point of upper limit of confidence interval to detect a significant difference in the number of individual animals bycaught if net profile is reduced in the experimental net.

Power analyses are commonly used to determine the sample size required to detect an effect between two treatments with a specified level of power. In general, field trials of gear modifications to reduce bycatch aim to detect a 50% reduction in bycatch with a power of 0.8 (Dawson et al. 1998). This requirement means that the development and testing of a specific modification will only be continued if it is shown to reduce bycatch substantially. Such power analyses are based on the assumption that, except for the specific gear modification, all other variables relating to fishing behaviour of the modified and unmodified fishing gear are the same.

The number of hauls needed to have a power of 0.8 to detect a 50% reduction in bycatch in an experimental net with a range of fishing areas relative to a control net was calculated using the power.of.sample function (Mike Lonergan, SMRU). This function allows a threshold to be set for magnitude of difference that is required in bycatch rates in experimental hauls to reject the null hypothesis of an insufficient difference in bycatch reduction. For example, specifying a threshold of <=0.5 requires a reduction of 50% or more in the experimental net. Changing this threshold to <=0.4 requires a reduction of 50% or more in the experimental net given the net had a 20% reduction in fishing height relative to the control net.

3.4   Results

3.4.1   Bridlington Trial

Each net was deployed and retrieved a total of five times, and soak duration of nets during deployments varied from 68 to 96 hours. Fish catches in all nets were recorded, but total catches were very low. The Single and Cigar net each caught a total of two thornback rays (*Raja clavata*) over the five deployments, whilst the Double net caught 1 thornback ray. In addition, the Single and Double net caught three and two mackerel, respectively. All thornback rays were released and no marine mammals were bycaught during the trial.

3.4.1.1   Are the fishing profiles of nets similar during deployments?

Figures 5 shows the range and mean float line heights recorded at ten minute intervals for each net during each deployment. The Double net fished with a higher mean float line height in all five deployments. The pattern was less consistent for the other two net types. The Single net had a higher mean float line height than the Cigar net in three of the five deployments, while the Cigar net had a mean higher float line height in the remaining deployments. The Cigar net was reported twisted when hauled at the end of the second deployment that resulted in the relatively low float line heights recorded. While the maximum float line height never exceeded the rigged height of 2m for either the Double or Cigar net, in deployments two and three the Single net did. Outliers in float line height data are likely a result of the +/0.40m accuracy in measurement of the depth sensors. Measurements close to or at zero reflect periods when the float line of the cigar net appeared to be lying on the seabed.
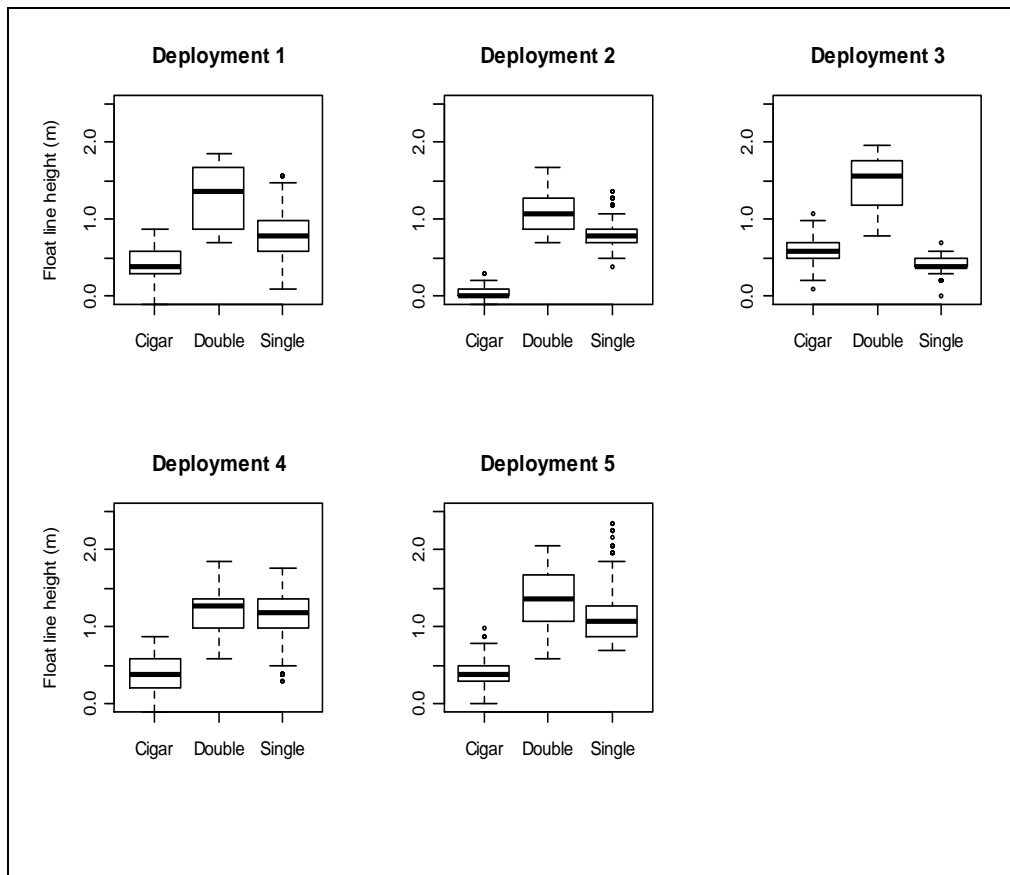
Figure 5 Box plot of float line heights recorded at 10 minute intervals for each deployment.

Table 4 summarises the mean float line height of each net for each deployment. Investigation of depth measurements recorded on the float line and lead line of the Cigar net indicate that the float line of this net did not fish properly during the second deployment and likely became twisted when shot, resulting in an average float line height of 0.03m for this deployment. During the third deployment, measurements recorded on the lead line of the Single net did not follow the same pattern as those recorded on the lead lines of the Double and Cigar net. Because depth sensors on the lead lines simply record the depth of water above them as it changes through the tidal cycle, the measurements of all three nets should be the same. The fact that measurements collected on the lead line of the single net did not follow this pattern suggests that the lead line of this net may have been lifting off the bottom, or the depth sensor was not functioning properly. These measurements resulted in the low mean float line height recorded for the Single net during the third deployment.

Net	Mean float line height (m)					Average
	1	**2**	**3**	**4**	**5**	
Double	1.30	1.11	1.45	1.22	1.37	1.29
Cigar	0.38	0.03	0.60	0.38	0.39	0.36
Single	0.79	0.81	0.40	1.14	1.12	0.85

Table 4: Mean float line height (m) recorded for each net type.

Figure 6 shows a plot of float line heights fishing over a representative 24 hour period in relation to predicted current speed and tidal height. The float line height of each net shows some periodicity in movement, however, increases and reduction in float line heights do not occur simultaneously in the three nets. Stewart & Ferro (1985) and Shimuzu et al. (1987) both reported a decrease in the float line height of gillnets with increased water current speed. The lowest float line heights recorded for the Single net and Cigar float net appear to follow current speeds predicted by POLPREDS reasonably well (Fig 6). However, the relationship between the Double net and predicted current speed is less clear, with the float line height being highest on one occasion during fastest predicted current speeds.
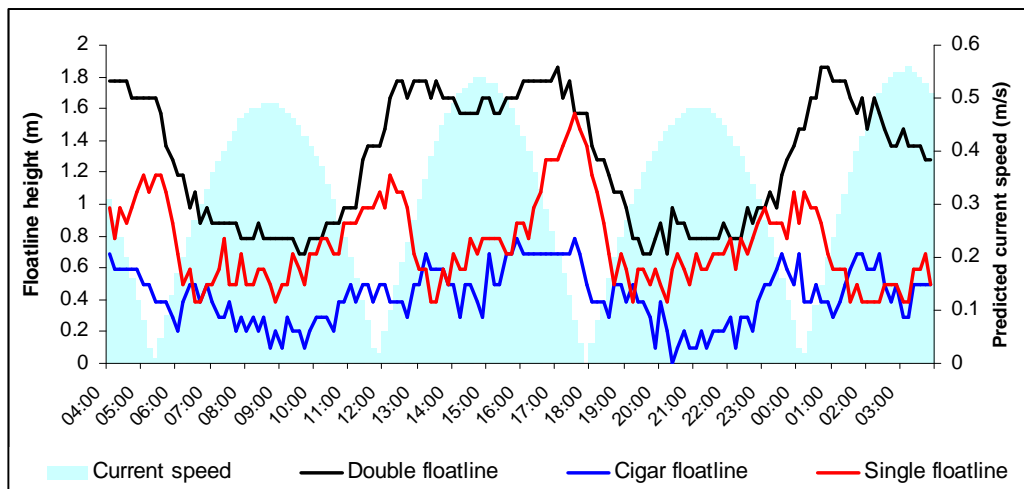
Figure 6: Calculated float line heights over a 24 hour period, and predicted current speeds.

A GLM with Gamma errors and an inverse log link function was constructed to investigate whether float line heights were different between the three nets. The only covariate retained by the model to predict float line height was net type. Means and confidence intervals of predicted net heights are summarised in Table 5.

Covariate	Parameter estimate	95% Confidence interval
Cigar net	0.34	0.2 to 0.46
Double net	1.34	1.29 to 1.39
Single net	0.87	0.78 to 0.96

Table 5: Estimates of mean float line height and 95% C.I.

Current speed was not retained as an explanatory variable for headline height for data collected at 6 hourly intervals. Therefore, simple GAMs were constructed to illustrate the relationship using data collected at hourly intervals. Figure 7 shows the relationship between float line height and current speed for all three nets.

Double net   Cigar net   Single net
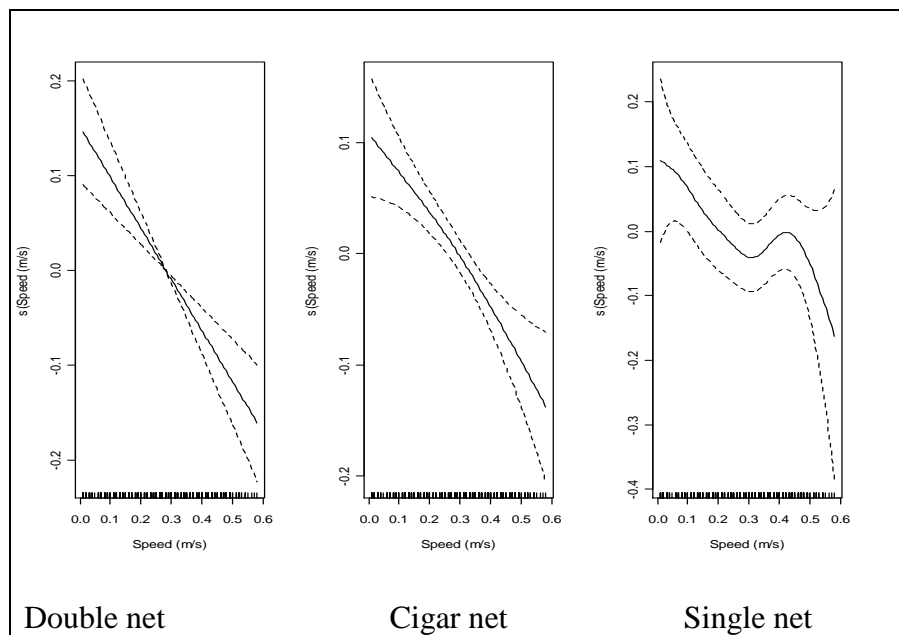
Figure7: Gam plots of relationship of float line height and current speed.

Therefore, the different amounts of flotation used in the three nets resulted in a significant difference in their relative fishing profiles, with the greatest difference in mean float line height being between the Double and the Cigar net.

3.4.1.2 Do individual nets behave the same way in each deployment?

While box plots of recorded float line heights showed that the double net fished with a higher profile in all deployments, the ranges of heights fished by this and the other two nets varied between deployments. In the bootstrapped Kolmogorov-Smirnov tests to investigate if the distribution of float line heights were significantly different for individual nets between deployments, the float line height of the Double net was

significantly different (p<0.05) for all pair wise comparisons of deployments with the exception of deployments 1 v 5 and 3 v 5. For the Cigar net, the distribution of float line heights was significantly different for all comparisons (p<0.05) with the exception of deployments 1 v 4 and 1 v 5. For the Single net all comparisons were again significantly different (p<0.05) with the exception of deployment 1 v 2.

Significant differences in float line height would be expected between hauls for individual nets if the environmental conditions, in particular current speed, varied greatly between deployments. Figure 8 shows the range of tidal heights and current speeds for each deployment predicted by POLPRED. With the exception of deployment 3, the range of tidal heights and mean current speeds were similar between deployments. Differences in distributions of float line heights between deployments might be attributed to the +/- 0.4m error in depth measurements recorded by the tags. However, because the same pairs of tags were used on each net for each deployment, the effect of this error is likely to have been minimised.
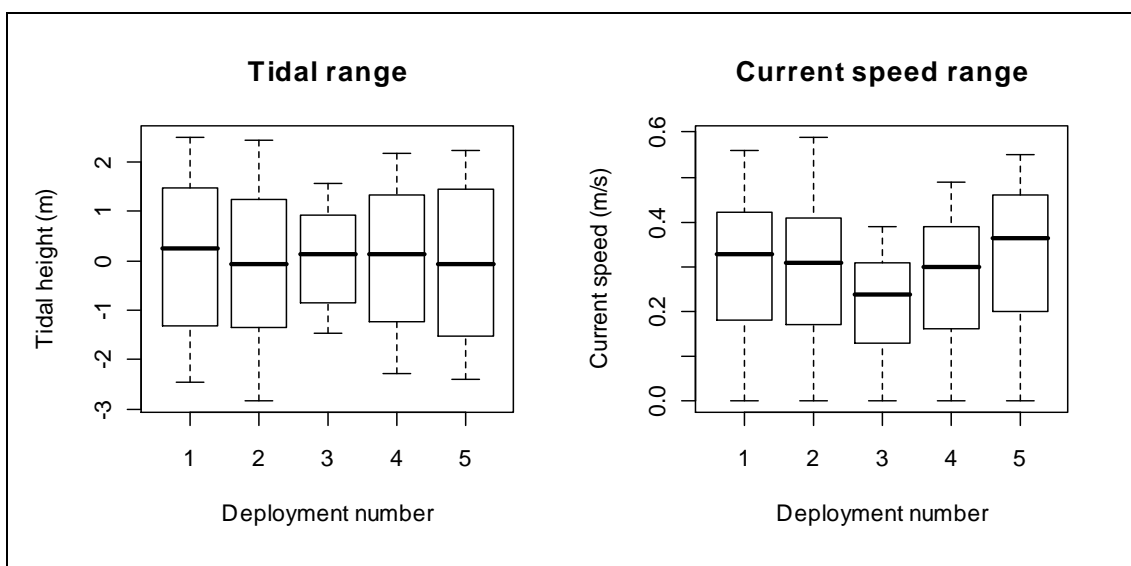
Figure 8: Range of tidal heights and current speeds predicted by POLPRED by deployment.

3.4.1.3 Rate of change of float line height

Density plots of the difference between the previous measurement and the subsequent measurement at consecutive hourly intervals are displayed in Figure 9. This plot shows that modal changes in float line height in consecutive hours were greater for

the Double net than the Single or Cigar net, although the tails of the distributions show that within an hour the change in the float line height of the Single net could be greater than 1m.
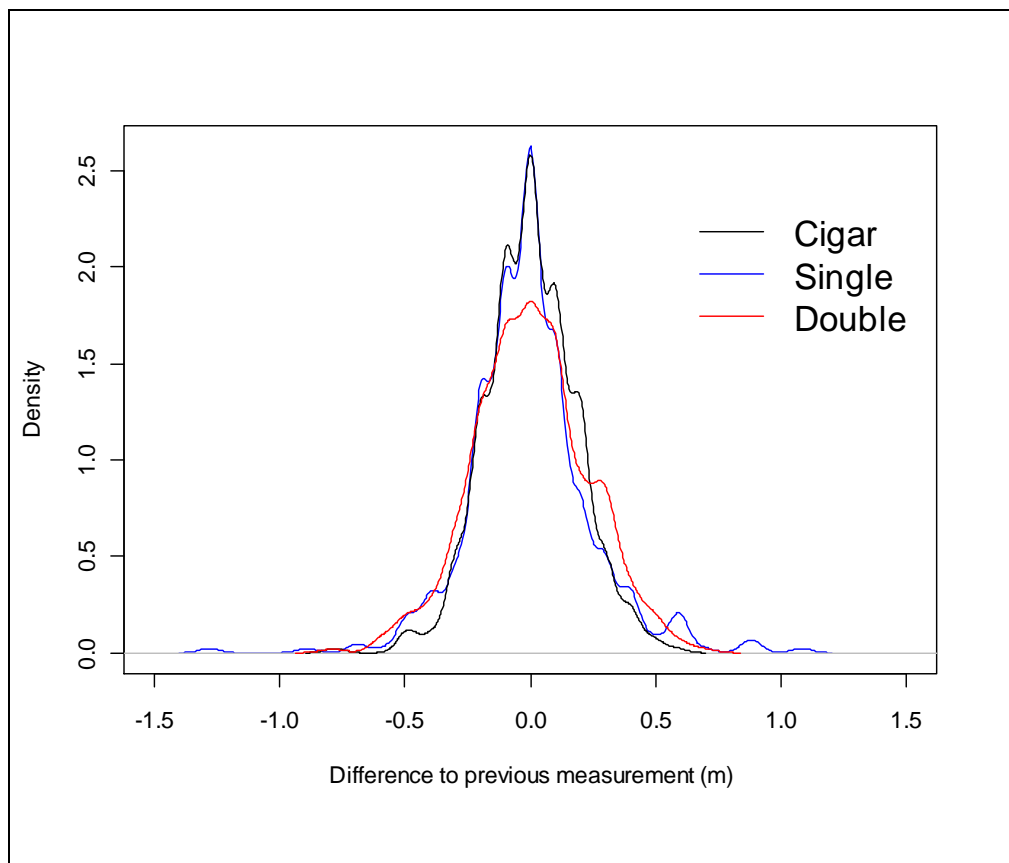
Figure 9: Density plots of difference to previous float line height measured at hourly intervals for all three nets over all deployments.

To investigate whether the float lines of pairs of nets fished the same amount relative to each other in different deployments, the difference in float lines was examined on measurements recorded at hourly intervals. Results of bootstrapped Kolmogorov-Smirnov tests showed that distributions of differences between the Single and Double nets were significantly different when all deployments were compared ($p<0.05$). Likewise, the distributions of differences between the fishing height of the Single and Cigar net were significantly different in all deployments. However, there was no significant difference between the Double and Cigar net between deployments 3 v. 4 ($p>0.05$) and 4 v. 5 ($p>0.05$). Figure 10 shows the distribution of differences in height measurements for pairs of nets for each deployment.
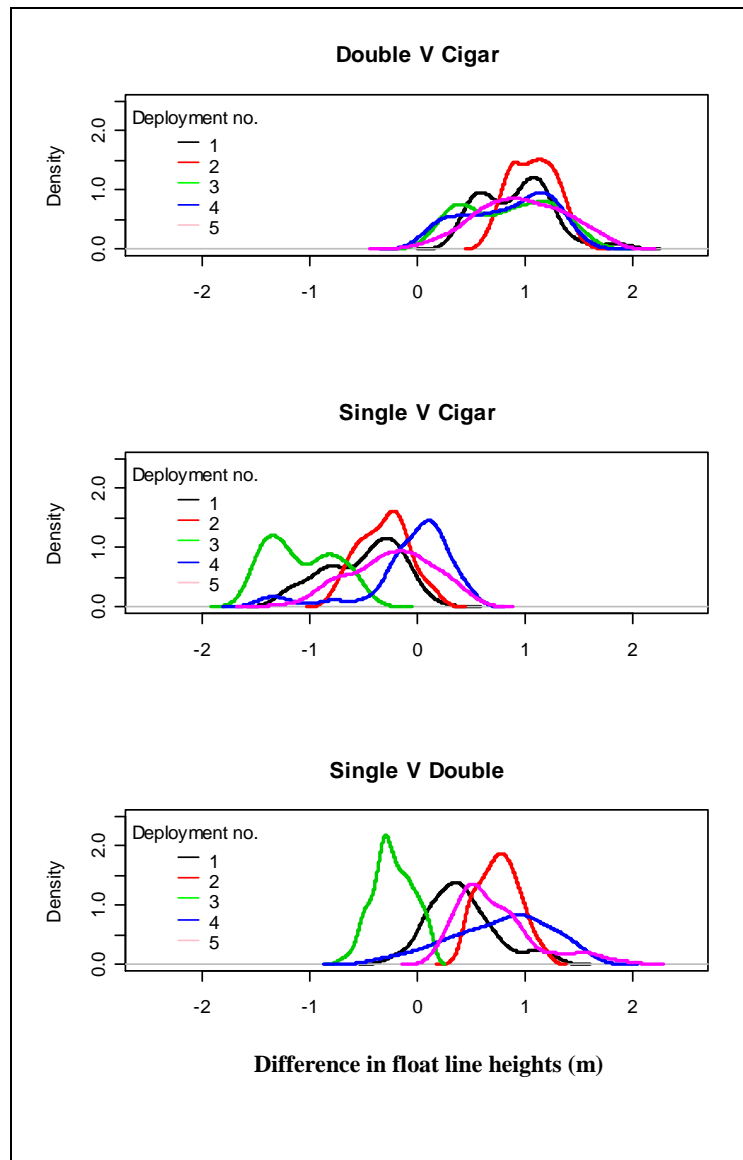
Figure 10: Distribution of difference in the float line height between pairs of nets. Data were bootstrapped 1000 times with replacement.

3.4.1.4 Proportion of total net area fished relative to rigged net height.

In order to assess the relative reduction in float line relative to the theoretical net area the proportion of theoretical area fished by each net were calculated. Figure 11 shows the distribution of the proportion of theoretical area fished by each net for all five deployments combined.
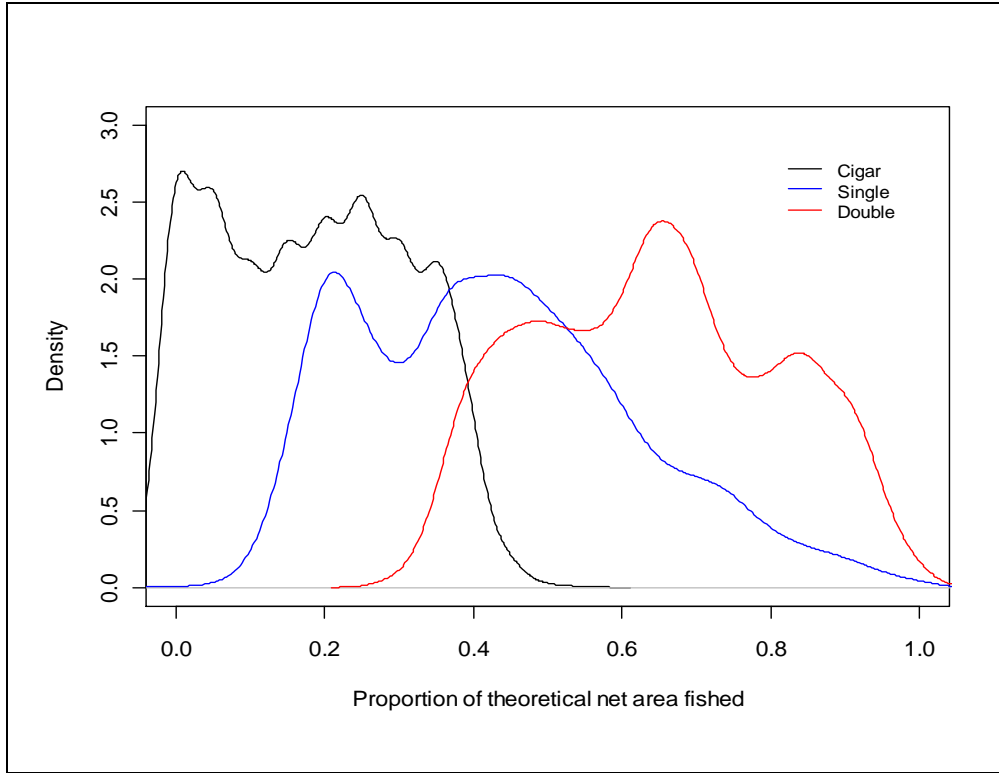
Figure 11: Proportion of theoretical net area of each net for all five deployments combined.

The average proportion of the theoretical two-dimensional area fished by the Double net was 0.65 over the duration of the five deployments. In contrast the Single net fished 0.43 of the theoretical net area while the Cigar net fished 0.19. For both these latter nets the average proportion fished for all deployments combined is reduced due to these nets not fishing properly during deployment 2 (Cigar net) and deployment 3 (Single net). Table 6 provides the average proportion of the theoretical net height fished for each deployment.

Deployment	Double	Single	Cigar
1	0.65	0.40	0.20
2	0.56	0.41	0.03
3	0.73	0.20	0.31
4	0.61	0.57	0.19
5	0.68	0.55	0.20

Table 6: Mean proportion of theoretical area fished per haul.

Excluding those deployments where nets did not fish properly, the average proportion of rigged height fished by the Single net was 0.48 and was 0.22 for the Cigar net.

3.4.2 St. Andrews Trial.

Two net deployments were conducted in St. Andrews Bay to compare the relative fishing heights of gillnets previously deployed in paired experimental trials. All three nets were first deployed on the 18th of September and retrieved on the 20th of September 2009; a total soak duration of approximately 48 hours. No marine mammals were caught during this first deployment, but a guillemot (*Uria aalge*) was bycaught in the bass net. Three mackerel (*Scomber scombrus)* were also caught in this net while no fish were caught in the standard skate net or $BaSO_4$ net. Nets were deployed for a second time between the 24th of September and the 27th of September 2009; a total soak duration of approximately 67 hours. No marine mammals were bycaught, but two guillemots were caught in the bass net, and one guillemot in the $BaSO_4$ net. No fish catches were recorded.

3.4.2.1 Are the fishing profiles of nets similar during deployments?

Figure 12 shows a box plot of float line heights of all three nets during the first and second deployments. The difference in the mean float line heights between the three nets reflects the difference in the rigged heights of these nets.
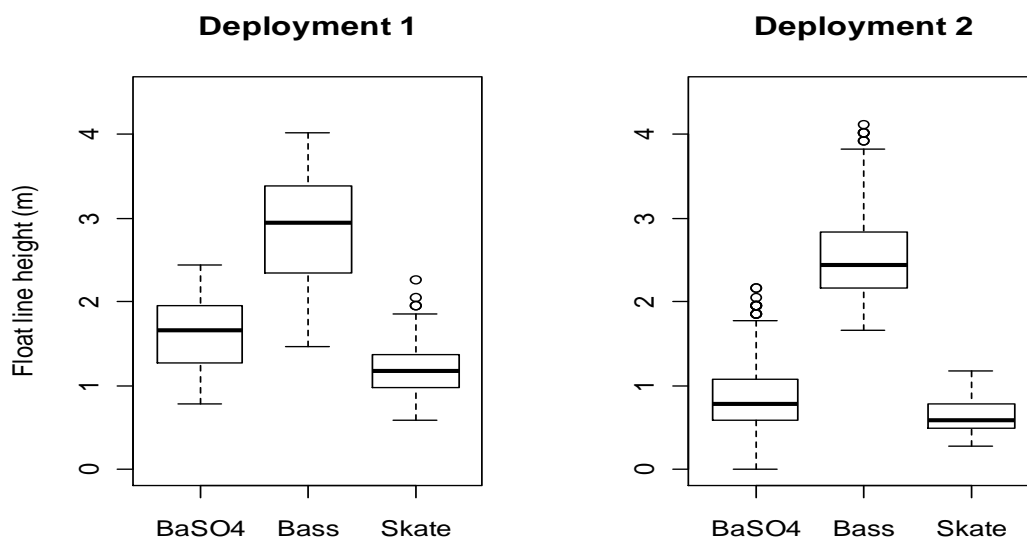
Figure 12: Box plot of float line heights recorded at 10 minute intervals for each deployment

However, the average float line height of all three nets was lower during the second deployment. This difference was greatest for the skate and $BaSO_4$ nets which fished with a mean float line height approximately half that of the first deployment. Mean float line heights for each net and each deployment are summarized in Table 7.

Net	Deployment 1	Deployment 2	Average
Skate	1.17	0.63	0.9
Bass	2.86	2.56	2.71
$BaSO_4$	1.62	0.87	1.25

Table 7: Mean float line height (m) recorded for each net type.

Figure 13 shows a plot of float line heights fishing over a representative period of 19 hours in relation to predicted current speed and tidal height. Changes in the height of the float line are relatively synchronous between the Bass and $BaSO_4$ nets while the relative changes in float line height of the Skate net are less pronounced. The relationship between float line height and current speed does not match the predicted trend of decreased float line height with increased current speed. This is likely a result of a poor prediction of current speeds by POLPRED in St. Andrews Bay where complex tidal currents exist due to a dominant anti-clockwise gyre caused by the ebb tide being more powerful than the flood tide.
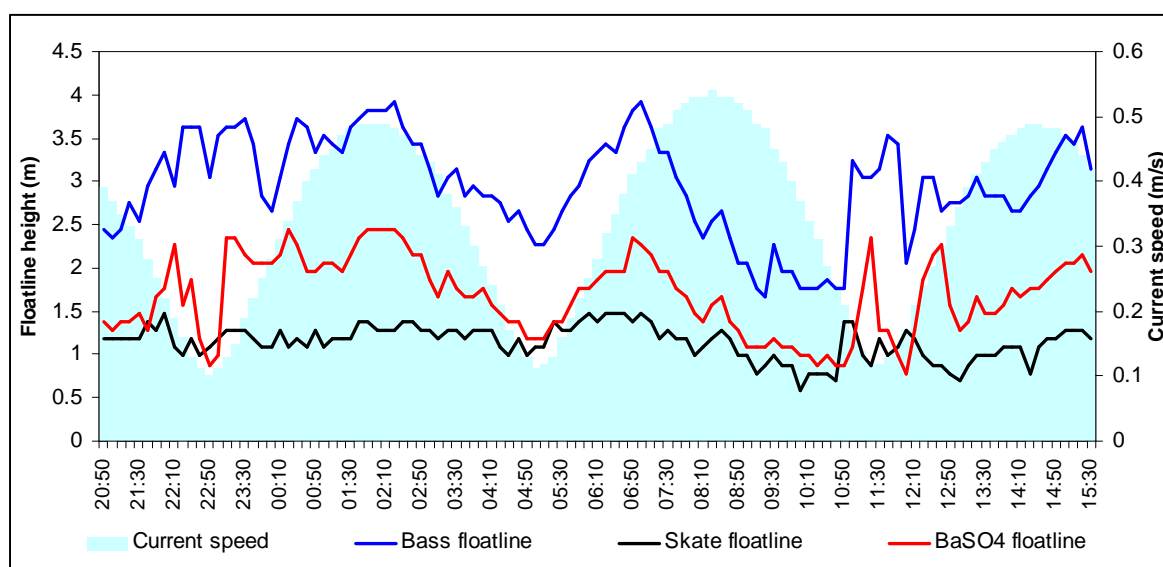
Figure 13: Calculated float line heights of three nets over a 24 hour period, and predicted current speeds.

A GLM with Gamma errors and an inverse log link function was constructed to investigate whether the float line heights of the three nets fished differently compared to each other. As all three nets had a different rigged height, dividing by the rigged height of each net standardized height measurements and these data were used as the response variable. To reduce the influence of autocorrelation on model results, data were restricted to measurements taken at the mid point of each tidal cycle that nets were deployed. Means and confidence intervals of predicted net heights are summarised in Table 8.

Covariate	Parameter estimate	95% Confidence interval	p-value
$BaSO_4$ net	1.33	1.13 to 1.61	P<0.001
Bass net	2.92	2.48 to 3.56	P<0.001
Skate net	0.92	0.78 to 1.12	P<0.001

Table 8: Estimates of mean float line height and 95% C.I.

Density plots of the difference between the previous measurement and the subsequent measurement at consecutive hourly intervals are displayed in Figure 14. These plots show that modal changes in float line height in consecutive hours were greatest for the Skate net and the $BaSO_4$ net in both deployments. For all nets, hourly differences in float line height were greatest during deployment 1.
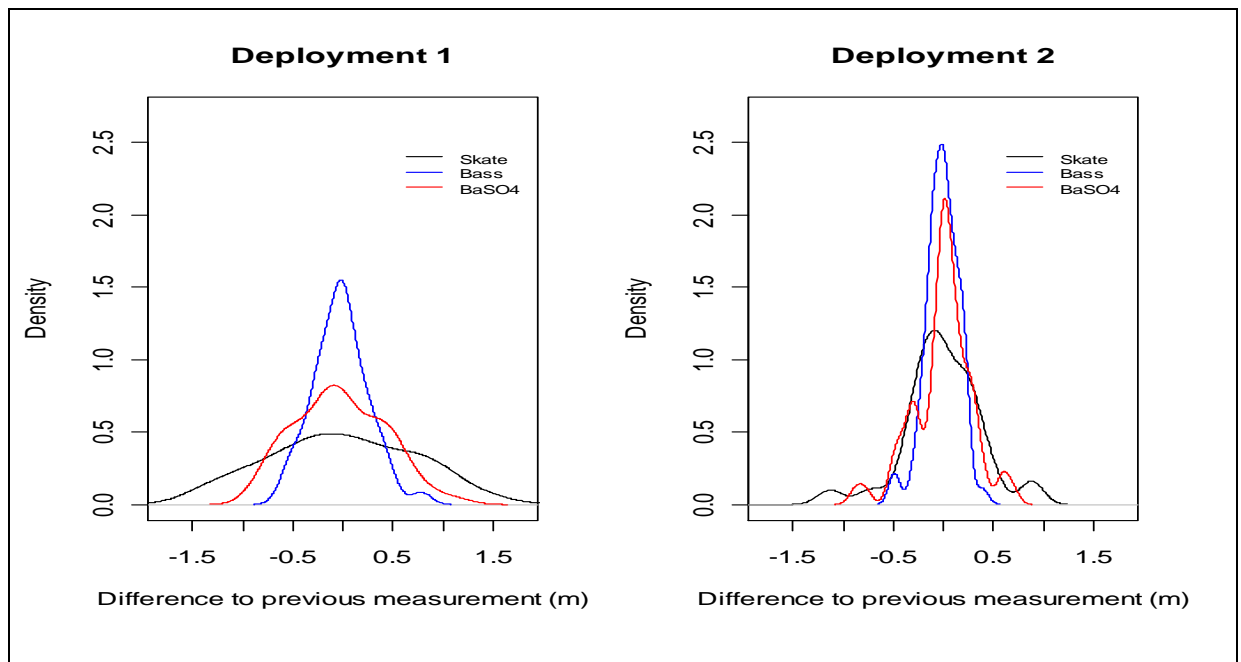
Figure 14: Density plots of difference to previous float line height measured at hourly intervals for all three nets over all deployments.

In addition, the distribution of the differences in float line heights for pairs of nets was significantly different for all nets between the first and second deployment (p<0.01).

3.4.2.2 Proportion of total net area fished relative to rigged net height.

The two dimensional area that each net fished relative to the theoretical rigged height was calculated for each 10 minute measurement of float line height. Figure 15 shows the distribution of the proportion of theoretical area fished by each net for the two deployments combined.
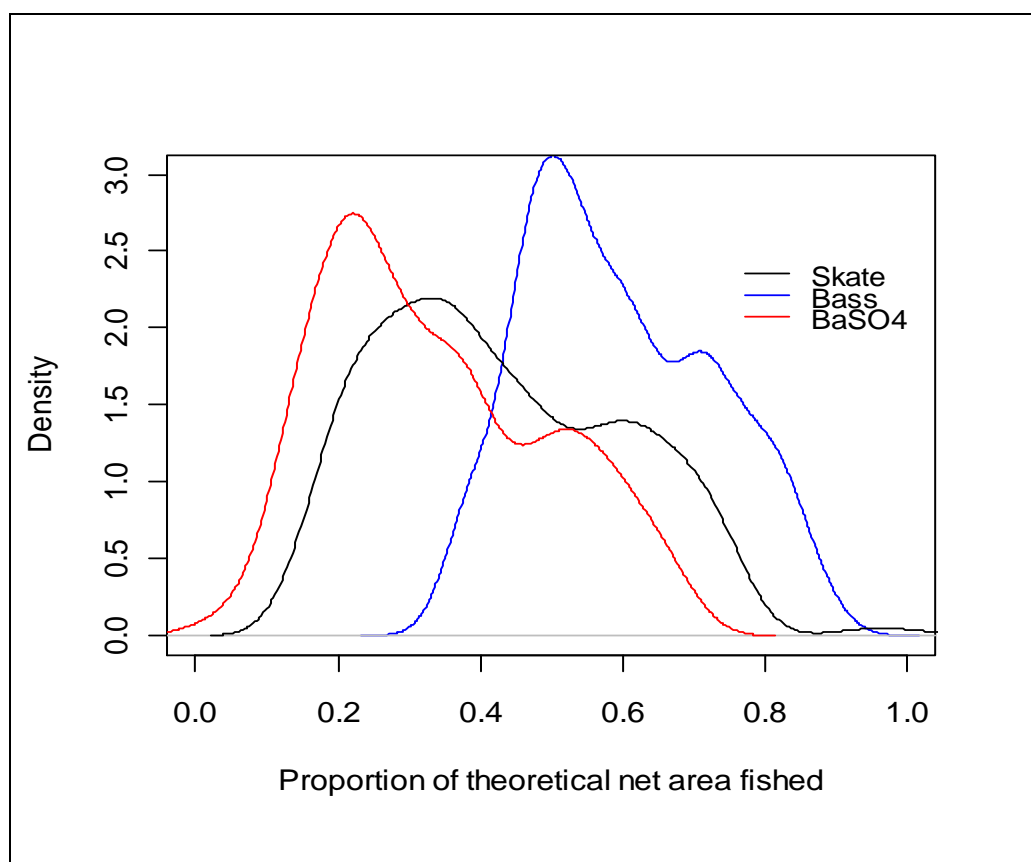
Figure 15: Proportion of theoretical net area of each net for all five deployments combined.

The average proportion of the theoretical two-dimensional area fished for both deployments combined was 0.60 for the Bass net, 0.45 for the Skate net and 0.36 for the $BaSO_4$ net. The Skate net had the same gear characteristics as the Single rigged net deployed in the trial in Bridlington, and the average proportion of the theoretical net area fished by the net was similar during both trials (0.43 v 0.45). Table 9 shows the average proportion of the theoretical net height fished for each deployment.

Deployment	Bass	Skate	BaSO$_4$
1	0.63	0.58	0.46
2	0.57	0.32	0.25

Table 9: Mean proportion of theoretical area fished per haul.

3.4.3   Effect of reduced net profile on significance and power

If the probability of cetacean bycatch is related to the active fishing profile of gillnets, then the significance of observed bycatch rates will vary dependent on whether the reduction in active fishing height or proportion of area fished is taken into account. Figure 16 shows the significance of different combinations of individuals caught in a control and experimental net, given a background bycatch rate of 0.04 animals per haul and a sample size of 200 hauls per net type. Black and red dots together indicate combinations of the numbers of animals caught in each net type, after 200 hauls, which would be considered significant if the profile of the experimental net was assumed to be the same as the control net. The subset of red dots indicate combinations of numbers of animals caught in the control net, relative to the experimental net, which would be considered significant after a change in net profile in the experimental net is accounted for.
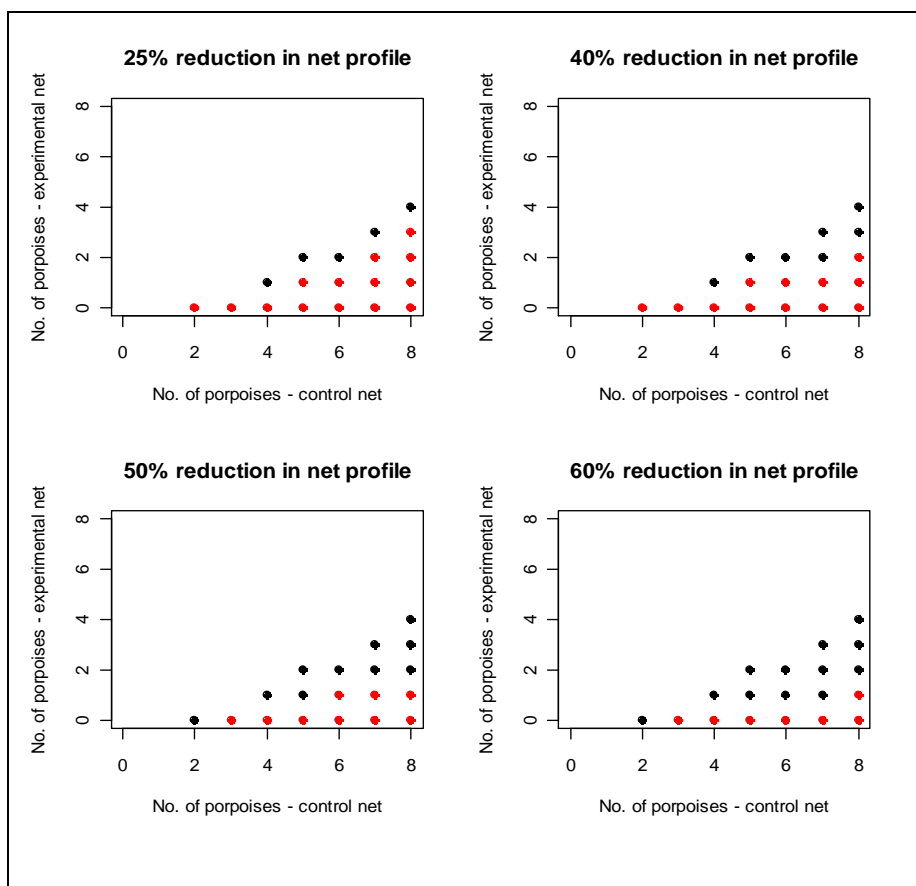
Figure 16: Combinations of animals caught in a control and experimental net after 200 hauls of each net type. Black dots indicate significant combinations when net profiles are assumed to be equal. Red dots indicate significant combinations after a reduction in net profile has been accounted for.

For example, an observed catch of eight animals in 200 hauls of the control net and four animals in 200 hauls of the experimental net would be considered a significant difference if net profiles were assumed to be equal. However, if the net profile of the experimental net was reduced by 25%, relative to the control net, these observed bycatch rates would not be significantly different. These results show that if the probability of cetacean bycatch is related to the fishing profile of a bottom set gillnet, then the power to detect significant differences between control and experimental nets will be affected if gear modifications inadvertently lead to a reduction in the fishing profile of the experimental net.

The number of hauls needed to be observed in a trial to have sufficient power to detect a required reduction of bycatch in modified gillnets, given an equal fishing profile of the control and experimental nets and a range of reductions in the net profile of the experimental net relative to the control net, was calculated. Assuming that the

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profile of a gillnet is linearly related to the probability of a cetacean being bycaught then any reduction in float line height due to a specific gear modification would need to be accounted for in order to determine that a given 50% reduction in bycatch is as a result of the actual gear modification and not the indirect effect this modification has on net profile. Figure 17 shows the number of hauls needed to be observed to detect a 50% reduction in bycatch given a range of different reductions in the float line height of a modified gillnet and a background bycatch rate of 0.04 animals per haul. If the control and modified gillnet fish with the same net profile, then 215 hauls of both the control and experimental net would need to be observed. If the profile of the modified net were indirectly reduced by 30% due to the gear modification then 302 hauls of each net would need to be observed to have the same power of detecting a 50% reduction in bycatch due to the gear modification. If the profile of the experimental net is reduced by 50%, then twice the numbers of hauls (n=440) are required than if the both nets were assumed to have the same fishing profile.
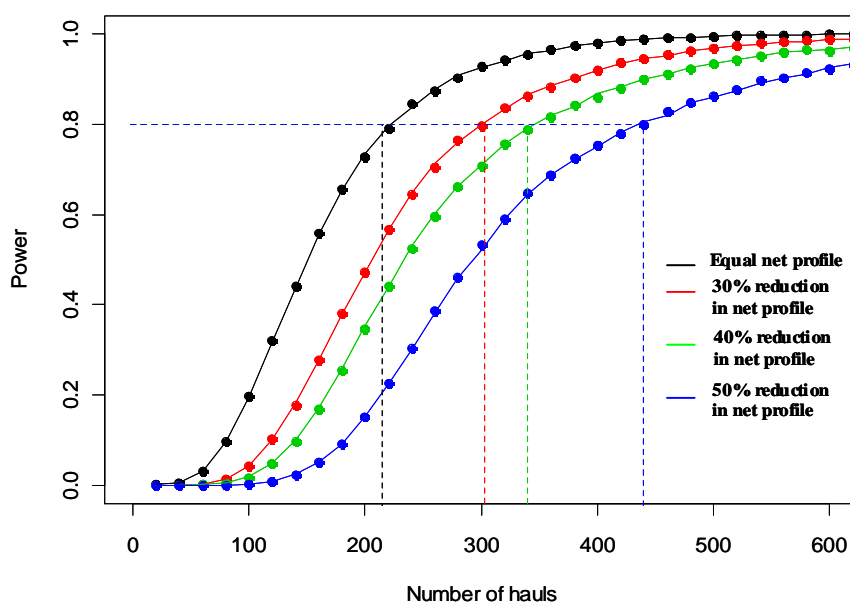
Figure 17: Dashed lines represent number of hauls needed to be observed in control and experimental nets to detect a 50% reduction in bycatch in the experimental net with a power of 0.8. Black dashed = equal net profiles in control and experimental net. Red dashed line – 30% reduction in profile of experimental net. Green dashed line – 40% reduction in profile of experimental net. Blue dashed line – 50% reduction in net profile of experimental net.

3.5 Discussion

It is unclear whether a relationship exists between the fishing height of bottom set gillnets and the probability of cetacean bycatch, although the use of tie downs, which effectively reduces the fishing profile of gillnets, has been related to reduced bycatch rates of harbour porpoises in the USA (Palka 2000) and common dolphins (ICES 2011). However, the relationship between cetacean bycatch rates and net profile have not been directly assessed for bottom set gillnets. Investigation of the factors affecting harbour porpoise bycatch in UK bottom set gillnets in ICES Area VII (Chapter 2) showed that higher bycatch rates occurred in gillnets with a rigged height of 3.8 to 6.2 metres. Measurements on the active fishing height of float lines of three bottom set gillnets in the experimental trials in Bridlington, showed that the proportion of net area fished, relative to calculated rigged net area, varied as a result of the type and amount of flotation used. Therefore, information on rigged net height as recorded by independent fisheries observers may not reflect the active fishing height of bottom set gillnets if other known gear characteristics are not considered.

3.5.1   Measured fishing heights of static gillnets

Results of comparisons of net heights from the trial in Bridlington showed that the relative fishing area of the skate net rigged with a polypropylene float line and cigar floats was on average 30% percent of the area fished by the skate net rigged with twice the amount of flotation, and 44% of the area fished by a standard skate net. If the probability of cetacean bycatch is related to the active fishing height of the gillnet, then any differences in net profile between control and modified nets in a pair trial may lead to wrongly inferring statistical significance to differences in catch rates.

For example, it is hypothesized that harbour porpoise may not detect gillnets at sufficient distances to avoid them, but detection ranges may be increased by increasing the target strength of elements of the net such as the float line. Consider a hypothetical trial to compare the bycatch rates of harbour porpoises in gillnets with single continuous float lines to gillnets with discrete ellipsoidal floats, using the same

gear characteristics as the Single and Cigar nets measured in the Bridlington trial. If, after observing 200 hauls in each net, eight harbour porpoises were caught in the control net (Single) and four in the experimental net (Cigar), we would conclude that the bycatch rates in the two nets were significantly different, and this significant result was due to the presence of the Cigar floats. However, if the reduction in net profile of the Cigar net (44% of the Single net) was accounted for then the catch rates of the two nets would no longer be significantly different. Of course, the active fishing height of gillnets may not be linearly related to the probability of cetacean bycatch, but the above example shows that unless all other parameters are equal, the results of paired trials with control and modified nets will be ambiguous. SMRU et al. (2001) found that harbour porpoise bycatch rates were significantly lower in nets with a standard polypropylene float line and additional plastic floats than in nets with a buoyant float line made of rope with a polystyrene core. Results from this chapter suggest that the reduction of harbour porpoise bycatch observed by SMRU et al. (2001) may have been due to the reduced fishing profile of the nets with cigar floats compared to those rigged with the buoyant float line with polystyrene core.

As well as the amount of buoyancy used, Stuart and Ferro (1985) found that twine area was a significant factor effecting netting drag, and therefore float line height, at different water current speeds. They calculated the twine area of one mesh as four times the knot-to-knot length of the mesh multiplied by the twine diameter. While rigged net height was retained as an explanatory variable of harbour porpoise bycatch in UK bottom set gillnet fisheries in ICES Area VII (Chapter 2), this result may have been driven by a lack of bycatches observed in sole gillnets which typically have a rigged net height of 1.6m. In addition to a lower rigged net height, the twine diameter of sole gillnets is also thinner than that used in taller profile gillnets targeting cod and hake.

The effect of twine diameter on harbour porpoise bycatch rates was investigated by Northridge et al. (2003) in a paired field trial. The two nets used in the study had the same gear characteristics as the bass net and standard skate net deployed with depth sensors in St. Andrews, although they were fished in longer fleet lengths. During this study significantly fewer harbour porpoise were caught in the bass net compared to the standard skate net. Results from the investigation of the float line heights of these

two nets deployed in St. Andrews showed the bass net fished with a mean float line height of 2.71m whilst the mean float line height of the standard skate net was 0.9m. If the probability of cetacean bycatch is related to the active fishing height of nets, higher bycatch rate would be expected in the taller bass net relative to the shorter skate net. However, during this trial the number of new holes (usually attributed by fishermen to be caused by seals) were counted in each net after each haul. Results showed that by the end of the trial the bass net had 19 more large holes than the skate nets, which the authors postulated may have been caused by harbour porpoise or seals becoming entangled in the net but then either breaking free or falling out of the net. If this postulation was correct then the results of this experiment would suggest that the probability of bycatch was similar for both nets regardless of fishing profile. However, as well as differences in fishing profile and twine diameter, the two nets in this trial also differed in mesh size, making it difficult to elucidate whether one or a combination of these factors resulted in the lower bycatch rates observed in the bass net.

Northridge et al. (2003) also investigated bycatch rates in standard skate nets and nets impregnated with $BaSO_4$ in a paired trial conducted between October 2002 and September 2003. Results of this trial showed no significant differences in harbour porpoise bycatch rates between the two nets. The mean float line height recorded by depth sensors for these nets were 1.24m for the $BaSO_4$ net and 0.9m for the skate net, therefore the profile of the skate net was 72% that of the $BaSO_4$ net. While these results would again suggest that net profiles do not influence harbour porpoise bycatch rates, the $BaSO_4$ net had a thicker and smaller mesh size than the standard skate net. Therefore, it is not possible to conclude which factors resulted in similar bycatch rates in the two nets.

A number of other studies have investigated bycatch rates of harbour porpoises in acoustically modified gillnets. Trippel et al. (2008) found significantly higher bycatch rates of harbour porpoises in standard gillnets than barium sulphate gillnets during paired field trials in Canada. Catch rates of target species were similar for the two nets with the exception of haddock (*Melanogrammus aeglefinus*) which were significantly lower in the barium sulphate net. The authors concluded that the reduction of harbour porpoise bycatch rates may be attributed to a combination of the increased acoustic

reflectivity of the barium sulphate net or the increased stiffness of the meshes (Cox and Read 2004). While barium sulphate nets have been shown to have a higher target strength than standard nylon nets (Trippel et al. 2003, Mooney et al. 2004, Mooney et al. 2007), the corresponding increase in detection distance for porpoises may not be large enough to give animals sufficient time to avoid entanglement (Mooney et al. 2007). Cox and Read (2004) investigated the echolocation behaviour of harbour porpoises in the vicinity of barium sulphate and standard gillnets and found no difference in echolocation rates recorded by PODs deployed with the two net types. No harbour porpoises were bycaught during this study, but the authors observed significantly more lobsters caught in the barium sulphate nets and postulated that the increased weight of the mesh material may have resulted in the modified nets lying on the sea floor for longer periods than the standard nets. They concluded that this difference in the fishing behaviour of the modified net may result in a lower probability of entanglement. Results from data collected on the active fishing height of a barium sulphate net in St. Andrews, showed that this net fished an average of 35% of its theoretical rigged height. Given the observations of Stuart and Ferro (1984) if additional flotation was not used to compensate for the extra weight and greater twine diameter of barium sulphate net material, then the net profile of a barium sulphate net would be expected to be lower than that of a standard net with the same rigged area, and the magnitude of this difference may increase with increased water current flow.

A paired trial comparing harbour porpoise bycatch rates in standard nets and nets impregnated with iron oxide also showed a significant reduction bycatch in modified nets (Larsen et al. 2007). However, additional flotation was added to the iron oxide nets in this trial to compensate for the increased weight of the net material. Visual inspection of the float line height of the iron oxide net and standard net in a flume tank indicated that both nets behaved in a similar manner under a range of water current flows. Therefore, the reduction of harbour porpoise bycatch observed in this trial cannot be attributed to a reduction in the fishing profile of the modified net. In addition, catch rates of cod were significantly lower in the iron oxide nets and the authors conclude that the reduction in both target catch and harbour porpoise bycatch was a result of the increased stiffness of the modified nets.

In summary, measurements of the float line height of the barium sulphate net deployed in St. Andrews Bay showed an average of a 65% reduction fishing area relative to the theoretical rigged area of this net. A significant increase in lobster catches in barium sulphate nets observed by Cox and Read (2004) suggests that the barium sulphate net used in their experiment had a reduced net profile relative to the standard net. While a reduced net profile may explain the reduction in bycatch rates of harbour porpoises observed by Trippel et al. (2008), Larsen et al. (2007) also observed a reduction of harbour porpoise bycatch in modified nets even though flume tank tests had shown that the additional flotation added to iron oxide nets had resulted in a similar net profile to the standard net. The results of Larsen et al. (2007) would suggest that reduced bycatch rates in these experimental nets is a result of increased stiffness of the barium sulphate and iron oxide twine relative to standard nylon. However, differences in catch rates of target species in these trials may indicate that two different mechanisms may have been involved in reducing harbour porpoise bycatch. Larsen et al. (2007) noted that more cod were captured by their gills rather than entangled in the iron oxide net and that, in general, captured cod were smaller than those caught in the standard net.

A study investigating the effect of increasing the gillnet tension (by increasing the amount of flotation and weight of the lead line) on shark bycatch rates found that modified nets reduced the bycatch of some species and the authors concluded that this was due to the reduced potential for sharks to become wrapped in the "stiffer" net material (Thorpe and Frierson 2009). Therefore, it is likely that the reduction in harbour porpoise bycatch in the Larsen et al. (2007) trial was a result of the increased overall stiffness/tension of the iron oxide modified nets. However, no significant difference in the bycatch rates of harbour porpoises was found during a paired trial using standard skate nets and skate nets rigged with double the amount of flotation (SMRU unpublished data). Results of an investigation into the mean float line height of each net showed that the standard rigged net fished approximately 66% of the area of the double net. Therefore, while the Double rigged net was likely to be more tensioned than the standard skate net, it also had a higher profile. In addition, the mesh size used in the trial by Larsen et al. was 156mm whereas the mesh size of skate nets is considerably larger. It is possible that larger mesh sizes affect harbour porpoise

bycatch rates more than other factors such as the amount of fishing profile or amount of tension of a net.

3.5.2 Potential effects on power

The results of paired trials to assess the potential for gear modifications to reduce bycatch rates of non-target species are difficult to interpret when more than one variable is changed. Therefore, it is important that the effects of gear modifications on the fishing behaviour of nets are assessed prior to the commencement of full-scale experimental trials. There are a number of issues that affect both the design and interpretation of trials to assess the effects of gear modifications on bycatch rates. The first of these is the cost and time required to run such trials. Because bycatch rates are generally low, trials must be conducted for a sufficient length of time to be able to detect any significant changes in bycatch rates between control and modified nets. In order to assess the number of hauls that need to be observed to do this, there is a requirement to know the underlying bycatch rates in control nets in the area where the experiment is to be conducted. These rates can then be used to conduct power analysis to determine the number of hauls required to provide sufficient power to detect a (specified) significant reduction in bycatch rates in modified nets. Unfortunately, if bycatch rates fluctuate between years, the rate of entanglement during a trial may be lower than that used to determine the number of samples required. However, any reduction in underlying bycatch rates may not become apparent until the end of the fishing season or experiment, at which point there may not be sufficient statistical power to detect a reduction in bycatch rates in modified nets.

Likewise if a modification of gear leads to an indirect reduction in the theoretical fishing area of the net (resulting in a decrease in the probability of bycatch occurring) and this is not taken into account when conducting a power analysis, then the number of hauls required to detect a specified reduction in bycatch, due to the specific gear modification, will be underestimated. In a fishery with a background bycatch rate of 0.04 bycatches per haul, a total of 215 hauls of each treatment would be required to in order to have 80% power to detect a 50% reduction in bycatch. If gear modification inadvertently led to a 30% reduction in the fishing profile of the modified net, and

therefore a 30% reduction in the probability of bycatch occurring, then 302 hauls of each net would need to be observed to retain a power of 80%. Therefore, it is important to understand how gear modifications change the way in which nets fish when designing experimental trials.

Self-contained depth recorders provide a methodology for investigating the behaviour of modified fishing gear (McFee et al. 2007, Brillant and Trippel 2009). Bycatch rates of large marine vertebrates are generally low and therefore require a large number of fishing operations to be observed in order to assess the usefulness of a specific gear modification. Without a clear understanding of how modifications alter the fishing behaviour of a specific gear, the results of such trials will at best be ambiguous, and at worst suggest that the specific gear modification, rather than an indirect unmeasured effect of this modification, is responsible for a reduction in bycatch rates. Results on the proportion of theoretical net area fished by the five different gillnets deployed with depth recorders show the importance of understanding how different gear characteristics will affect the fishing profile of the net. These effects will be amplified if nets are set perpendicular to the current, or in areas with higher current speeds (Stewart 1988). Therefore, the results of a gear trial in one area may not translate to another fishing area if tidal currents are greatly different. In order to maximize the utility of bycatch reduction trials and allow results to be tested and compared in other locations then the effects of gear modifications on experimental nets relative to control nets need to be examined in parallel to any bycatch mitigation trials.

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4. Chapter 4: Investigating the underwater fishing behaviour of standard and modified gillnets in a bycatch mitigation trial.

4.1 Abstract

Depth loggers were used to investigate the underwater fishing behaviour of one standard and two modified gillnet types in a bycatch mitigation trial in San Clemente, Argentina. Results showed that the $BaSO_4$ net fished with a significantly lower mean float line height than either the control or Stiff net. A generalized linear model (GLM) was used to determine which factors were related to Franciscana dolphin (*Pontoporia blainvillei*) bycatch rates during this trial. There was no significant difference in Franciscana bycatch rates between the three nets, or in the CPUE of target catch. These results suggest that reducing the fishing profile of nets may not provide a ubiquitous solution of reducing small cetacean bycatch in static gillnets. The only significant predictor of Franciscana bycatch was latitude. While acoustically modified nets have previously been shown to significantly reduce the bycatch rates of harbour porpoises (*Phocoena phocoena*) results of this study indicate that they do not hold much promise in reducing the bycatch rates of Franciscana dolphins.

4.2 Introduction

The density of nylon is similar to that of seawater (Pence 1986 cited in Larsen et al. 2007) and for this reason it has been hypothesized that odontocetes are unable to acoustically detect bottom set gillnets at sufficient distances to avoid them. Strategies to increase the detectability of gillnets to cetaceans have utilised pingers or have aimed to increase the acoustic reflectivity of gillnets to cetacean echolocation clicks. Although pingers have been shown to reduce bycatch rates of a number of cetacean species including harbour porpoises (*Phocoena phocoena*) (Kraus et al. 1997, Trippel et al. 1999, Gearin et al. 2000, SMRU et al. 2001, Larsen et al. 2002), Franciscana (*Pontoporia blainvillei*) (Bordino et al. 2002), beaked whale species (Carretta et al. 2008), short-beaked common dolphins (*Delphinus delphis*) (Barlow and Cameron 2003), there has been little to no implementation of these devices in fisheries outside of the USA or Europe. Reasons for low uptake of pingers include the high cost of these devices and the difficulty and cost in monitoring and enforcing their use. A

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number of studies have therefore aimed to reduce cetacean bycatch in bottom set gillnets by increasing the acoustic reflectivity of nets. Initial trials tested a number of modifications including the addition of passive reflectors, braided rope, air-tube nylon threads or multi-filament threads (Dawson 1991, Dawson 1994, Hatakeyama et al. 1994, Silber 1994, Koschinski 1997), but results of these studies were mixed.

In the late 1990s Trippel et al. (2003) tested a modified bottom set gillnet, where the density, and thereby acoustic reflectivity of the nylon, was increased through the addition of barium sulphate particles ($BaSO_4$). The results of this trial showed a significant decrease in harbour porpoise bycatch rates in the $BaSO_4$ net (Trippel et al. 2003). Larsen et al. (2007) also found a significant decrease in harbour porpoise bycatch rates in a net that had been made more reflective by the addition of iron Oxide (IO). However, they also observed a significant decrease in catch per unit effort (CPUE) of cod (*Gadus morhua*) in the IO nets. The authors concluded that the decrease in the bycatch of harbour porpoises was likely due to the increased stiffness of the iron oxide net. In contrast, a trial of standard and $BaSO_4$ nets by Northridge et al. (2003) found no reduction in harbour porpoise bycatch in the modified nets. However, the mesh size and rigged height of the control and experimental nets used in this trial were not equal, therefore making it difficult to elucidate which factors most contributed to the observed bycatch rates. In 2009 Trippel et al. (Trippel et al. 2009) reanalysed data collected from their first trial with additional data collected in subsequent years and again reported a reduction in the bycatch of harbour porpoises in $BaSO_4$ nets compared to control nets. This second analysis also showed a significant reduction of haddock (*Melanogrammus aeglefinus*) catches in the modified nets, but no significant difference in catch rates of Atlantic cod, pollock (*Pollachius virens*) or spiny dogfish (*Squalus acanthias*).

The acoustic properties of these "reflective" nets have been tested in both field and laboratory trials (Trippel et al. 2003, Mooney et al. 2004, Koschinski et al. 2006, Larsen et al. 2007, Mooney et al. 2007). Trippel et al. (2003) found that $BaSO_4$ gillnets were around three times more reflective than standard nets when ensonified with a 200 kHz multibeam sonar. Mooney et al. (2004, 2007) using generated broad band dolphin-like clicks and narrowband porpoise-like clicks, reported that the target strength of both $BaSO_4$ and IO nets was greater than comparable nylon nets at, or

near, perpendicular angles. However, the returned levels of echolocation click signals was the same for both reflective and unmodified nets when the angle of incidence was greater than 40 degrees. Mooney et al. (2007) also found that although iron oxide nets had a higher density than barium sulphate nets, they had lower relative target strengths (TS). In contrast, a separate study using sound pulses of 140 kHz 200μs at 140 kHz at a distance of 2 m from an IO and standard net found no significant difference in target strength between the two net types (Larsen et al. 2007). In addition, a field trial which utilised self-automated porpoise click detectors (TPODs) to examine the echolocation behaviour of wild harbour porpoise around reflective nets found no difference in echolocation rate or echolocation intensity compared to control nets (Cox and Read 2004).

The observation of no significant increase in the echolocation rate of harbour porpoise around acoustically reflective nets, and the increased stiffness of both $BaSO_4$ and IO nets relative to standard nylon nets has led a number of authors to postulate that observed reduction in harbour porpoise bycatch rates may have been as a result of the increased stiffness of the modified nets (Cox and Read 2004, Larsen et al. 2007, Trippel et al. 2009). Therefore, an international project (funded by the Lenfest Ocean Program) was initiated to examine bycatch rates of small cetaceans in standard, $BaSO_4$ and chemically stiffened nets, in a number of static gillnet fisheries around the world. One of these trials was conducted in an artisanal static gillnet fishery in Argentina where there is a high bycatch of Franciscana dolphins (*Pontoporia blainvillei*).

This species is endemic to the south-western Atlantic Ocean, ranging from Itaúnas, Brazil (18° 25'S) to the Province of Rio Negro, Argentina (41° 09'S) (Bordino et al. 2002), although it does not appear to be continuously distributed throughout its range (Secchi et al. 2003). Franciscana dolphins feed mainly on bottom dwelling fish species (Secchi et al. 2003) and data from satellite tags indicate dispersal of a maximum of 20km from the site at which they were captured (Bordino and Wells cited in Mendez et al. 2008). Bycatches of this species in gillnets have been reported throughout their coastal range (Praderi 1989, Crespo et al. 1994, Bordino et al. 2002, Secchi et al. 2003), and the Franciscana is considered to be the most threatened cetacean species in this region.

The first aim of this chapter was to investigate the underwater fishing behaviour of standard, $BaSO_4$ and "stiffened" gillnets and to see how fishing behaviour of each net related to observed bycatch rates. The second aim of this chapter was to use statistical models to assess which factors were related to the bycatch of Franciscana dolphins.

4.3   Materials and Methods

4.3.1   Fishing trials

The study was conducted in the waters of Bahia Samborombon and offshore of San Clemente del Tuyu, Buenos Aires Province, Argentina. Data were collected aboard four fishing boats from a local artisanal bottom set gillnet fishery consisting of approximately 50-60 boats. The fleet consists of 6-10m fibreglass launches with outboard engines between 50-125 hp. The main target species of the fishery are striped weakfish (*Cynoscion guatucupa*) and white croaker (*Micropogonias furnieri*). Two fishing boats operated predominantly in waters east of San Clemente where the maximum depth of net deployment was 17 metres, while two operated predominantly within Bahia Samborombon, one from San Clemente port the other further north from Canal 15. These boats fished in depths of 4-5 metres at low water. A fifth fishing boat was used during a second trial conducted in August, 2010. A map of fishing locations is provided in the results section.

The main aim of the trial was to investigate bycatch rates of Fransiscana dolphins in three different types of net, a control net, a chemically stiffened net and a net impregnated with $BaSO_4$. Each net consisted of a 100m long panel of 140mm stretched nylon monofilament net, with a rigged height of 3.5m. All nets were rigged with equal amount of flotation and lead line weight. Table 1 summarises the gear specifications of the three nets utilised in the trial. Each net had a unique number code and the type of net was identifiable by a coloured buoy at either end of the net.

Net Type	Mesh size	Net length	Rigged net height
Control	140mm	100m	3.5
Stiff	140mm	100m	3.5
$BaSO_4$	140mm	100m	3.5

Table 1: Gear specifications of control and experimental nets.

The nets were hauled by hand, with both the headline and lead line brought aboard the boat (Fig 1). The fishermen then moved along the length of the net removing fish. The anchors at either end of the net were not hauled and stayed in the same position whilst the net was deployed. Depending on the amount of fish catch, it took between 10 and 60 minutes to process a single net. An independent observer was present during each haul and recorded information on fish catches and discards, environmental variables and dolphin bycatches. All bycaught dolphins were returned to port and necropsied.

Figure 1: Hand hauling and processing of a gillnet.

4.3.2 Collection of float line height data.

A pair of Star ODDi DST-milli© depth temperature loggers (www.star-oddi.com) was deployed on the float line and lead line of each net following the methodology described in Chapter 3 during two experimental field trials in San Clemente de Tuyu, Argentina. Depth loggers were attached using cable ties and string and were placed approximately in the middle of each net. The middle of the net was calculated by counting the number of floats on the headline of the net. Depth loggers were deployed for either 48 or 96 hours before retrieval. For each net with tags deployed additional data on fish catches and environmental conditions were collected by onboard observers. Each logger was set to measure depth at 10 minute intervals. During the first trial it became apparent that some of the depth sensors were not recording properly and as a result the manufacturers replaced three of the sensors. On

completion of the trial all 9 sensors were returned to the manufacturers for pressure testing, so as to obtain an offset value for each sensor in order to standardize the data. However, these offset values did not provide consistent results and although a number of other methods to standardize the data were tried, it was clear that errors remained and the data were too inaccurate to draw conclusions on the comparable fishing behaviour of the three nets. The manufacturers replaced the original six sensors and six more were purchased for a second field trial. Prior to this second trial the sensors were pressure tested in St. Andrews and tested again in the field.

4.3.3   Calculating float line height – First trial.

The six tags deployed during this trial had all recorded accurately when deployed in a previous trial in the UK (Chapter 3) and results from the first deployment in San Clemente also appeared to be accurate. However, during the second deployment in San Clemente it was clear that at least two of the tags were not recording properly. Therefore, tags were calibrated twice in a pool approximately 3 metres deep at Mundo Marina, San Clemente. The results of these calibrations showed that some tags had a measurement error greater than $\pm$ 0.4m as specified by the manufacturer. Results of these calibration tests were sent to the manufacturer who stated the tags had gone through a pressure shift, they then replaced three of the tags and offered to calculate offset values for all tags once the trial was completed. However, when data were reanalysed using these offset values it was clear that some of the offsets overestimated the difference from the recorded depth measurement to the "real" depth measurement. This was apparent when some calculated net heights were greater than the actual heights of the nets. Table 2 shows the difference in the offset values provided by the manufacturers to those calculated from the first calibration in the pool at Mundo Marina.

	A	B	C	D	E	F
Tag number	10426	10427	10428	11261	11264	11265
Mean depth measurement in pool (bar)	0.33	0.25	0.31	0.4	0.09	0.13
Manufacturers offset	**-0.07**	**+0.01**	**-0.04**	**-0.05**	**+0.3**	**+0.21**
Adjusted depth measurement (bar) using manufacturers offset	0.26	0.26	0.27	0.35	0.39	0.34
Offset calculated from mean pool measurements	**-0.07**	**+0.01**	**-0.04**	**-0.14**	**+0.17**	**+0.13**
Adjusted depth measurement (bar) using offset calculated from pool data	0.26	0.26	0.26	0.26	0.26	0.26

Table 2: Summary of offset value for each tag provided by the manufacturer and calculated from the first tag calibration in a 3m pool.

While the offset values obtained from the Manufacturer and the pool calibration agreed for tag A and B and were within 1 bar for tag C, there was a large disagreement for the other three tags. This is likely to have been due to a second or continued pressure shift for these tags, and it is impossible to know exactly the shift in these tags happened. Therefore differences in depth measurements recorded by tags placed on the lead lines of nets during each deployment were compared. As nets were shot in similar depths within deployments depth measurements from these tags should be similar. Where measurements were not similar the effect of using calculated offset values was assessed. If an offset value for a specific tag improved the similarity in depth recordings measured by the tags deployed on lead lines within a deployment then this offset value was used for this tag until a deployment where a larger offset (i.e. that provided by the manufacturer) reduced the difference in lead line depth measurements. However, the graphs produced using these offset values (Appendix 1) are purely illustrative of how the three nets fished relative to each other and therefore cannot be used to make a quantitative comparison of net fishing heights.

4.3.4   Calculating float line height – Second trial.

All tags were pressure tested in St. Andrews, Scotland prior to the commencement of the second trial. Tags were found to be within $\pm$ 0.4m accuracy described by the

manufacturers. These data were used to group tags into pairs, where each pair contained two tags with most similar depth readings from the pressure test. Pairs of tags were then deployed on the lead line of a single net for 24 hours. Figure 2 illustrates pairs of tags attached to a single lead line. Data were downloaded and the average difference in depth readings between depth loggers in each pair was calculated. This average difference was then used as an offset value for each pair in order to improve the accuracy of measurements of float line height. Each pair of depth sensors was then rotated in turn between the three net types deployed during the second trial.

Figure 2: Deployment of tag pairs on lead line

4.3.5   Data analysis

4.3.5.1   Comparing float lines

Comparison of the active fishing height of the three net types were conducted following the methodology outline in Chapter 3.

4.3.5.2   Modeling observer data

The methods used to model the observer data are outlined in Chapter 2. To investigate which factors are related to dolphin bycatches in this fishery a GLM with Poisson error distribution was constructed. Candidate explanatory variables are listed in Table 3. The soak duration of each net was included in the model as an offset.

Explanatory variable	Description
Net type	Standard, BaSO4 or Stiff net
Latitude	Decimal degrees
Longitude	Decimal degrees
Net orientation	Set parallel or perpendicular to the current
Fish catch	Total catch per unit soak time (kg/hr)
Boat	Identification of vessel fishing
Net number	Numerical identifier of each net.

Table 3: Candidate explanatory variables.

4.4   Results

4.4.1   Investigation of float line height – first trial

Data on the float line heights of the control, Stiff and $BaSO_4$ nets were collected on seven fishing trips between the 26th of October and 17th of December 2009. Table 4 provides a summary of all tag deployments during the first experimental trial.

Deployment number	Boat	Haul no.	Date in	Date out	Approx. depth (m)	Net orientation to current	Total days recording
1	A	1	26/10/2009	28/10/2009	7	Parallel	2
1	A	2	28/10/2009	30/10/2009	7	Parallel	2
2	B	1	01/11/2009	03/11/2009	15	Parallel	2
2	B	2	03/11/2009	04/11/2009	15	Parallel	1
2	B	3	04/11/2009	05/11/2009	15	Parallel	1
3	A	1	10/11/2009	12/11/2009	7	Perpendicular	2
4	A	1	15/11/2009	17/11/2009	7	Perpendicular	2
4	A	2	17/11/2009	20/11/2009	7	Perpendicular	3
5	B	1	22/11/2009	23/11/2009	8	Parallel	1
5	B	2	23/11/2009	26/11/2009	8	Parallel	3
6	C	1	29/11/2009	30/11/2009	6	Parallel	1
6	C	2	30/11/2009	02/12/2009	6	Parallel	2
6	C	3	02/12/2009	04/12/2009	6	Parallel	2
6	C	4	04/12/209	05/12/2009	6	Parallel	1
7	D	5	09/12/2009	17/12/2009	5	Parallel	8

Table 4: Summary of all deployments during the first trial

Given inaccuracies in depth logger measurements data from the first trial were not suitable for quantitative analysis. However, illustrative graphs of the fishing behaviour and relative float line heights of the three nets, during each deployment, are provided in Appendix 1.

4.4.2 Investigation of float line height second trial.

Data were downloaded from pairs of tags deployed on a single lead line for 24 hours in a depth of 4m. The average difference between depth readings between sensors in each pair was calculated and used as an offset value for each pair (Table 5).

Pair	A		B		C	
Sensor	12510	12512	11870	11869	12513	11868
Offset	+0.1	0	0	+0.2	+0.4	0

Table 5: Offset in metres used for each pair of tags.

A pair of depth sensors was deployed on the float line and lead line of each of the three net types during three fishing trips conducted between the 7th and 19th of August, 2010. Each pair of tags was rotated between nets after each deployment to reduce any effect of inter tag variability on recorded float line heights. Table 6 provides a summary of deployments during the second trial.

Deployment number	Boat	Haul no	Date in	Date out	Approx. depth (m) at low water	Net orientation to current	Total days recording
1	A	1	07/08/2010	09/08/2010	4	Perpendicular	2
2	A	2	09/08/2010	15/08/2010	4	Perpendicular	6
3	B	1	15/08/2010	19/08/2010	4	Perpendicular	2

Table 6: Summary of deployments during the second trial.

Figure 3 shows the range and mean float line height recorded for each net during the three deployments. In all deployments the float line of the control and Stiff net fished at similar heights, whilst the $BaSO_4$ net fished with a mean height between 0.4 and 0.6m lower than the other two nets.
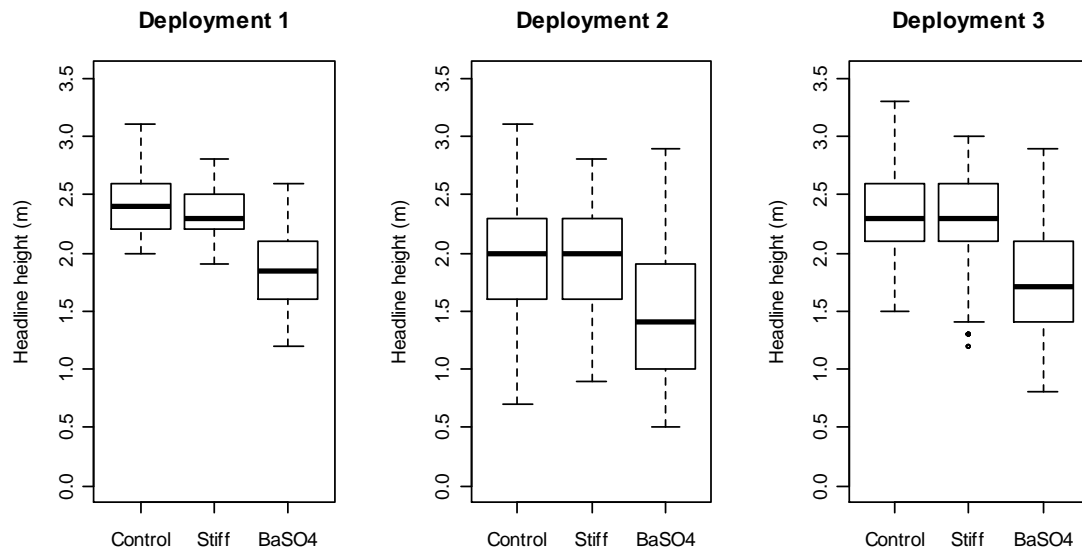
Figure 3: Box plots of float line heights in metres recorded at 10 minute intervals.

Results of a GLM model with Gamma error distribution and inverse link function showed that the float line height of the $BaSO_4$ net was significantly lower than the control net, while there was no significant difference in float line height between the control net and the Stiff net (Table 7).

	Estimate	Std. Error	p-value
(Intercept)	0.81	0.046	<0.001
BaSO4	-0.20	0.058	<0.001
Stiff	-0.02	0.065	>0.05

Table 7: Output of GLM (Gamma error distribution with inverse log-link function.

Mean float line height recorded for the control net was 2.26m (95% C.I.: 2.07-2.17), 2.23m for the Stiff net (95% C.I.: 2.05-2.14) and 1.84m for the $BaSO_4$ net (95% C.I.: 1.17-1.77).

The measurements of float line height recorded at 10 minute intervals were used to calculate the proportion of area each net fished relative to the rigged height of the net. During the first trail the Standard net fished 70% of the rig net area on average, the Stiff net fished 67% and the $BaSO_4$ net fished 53%. During the second trial the Standard net fished 55% of the rig net area on average, the Stiff net fished 56% and

the $BaSO_4$ net fished 43%. During the third trial the Standard net fished 67% of the rig net area on average, the Stiff net fished 66% and the $BaSO_4$ net fished 50%.. The lower average area fished, relative to the rigged net area, by all three nets during the second deployment are due to the change in behaviour of these nets over the course of the 6 day deployment when the nets could not be retrieved because of strong winds.

While the difference in the mean float line height of the $BaSO_4$ net relative to the Standard and Stiff net indicates that the former generally fishes with a lower profile than the other two nets, differences between nets were not consistent between hauls. Figure 4 illustrates the distribution of differences in float line height between pairs of nets during each deployment.
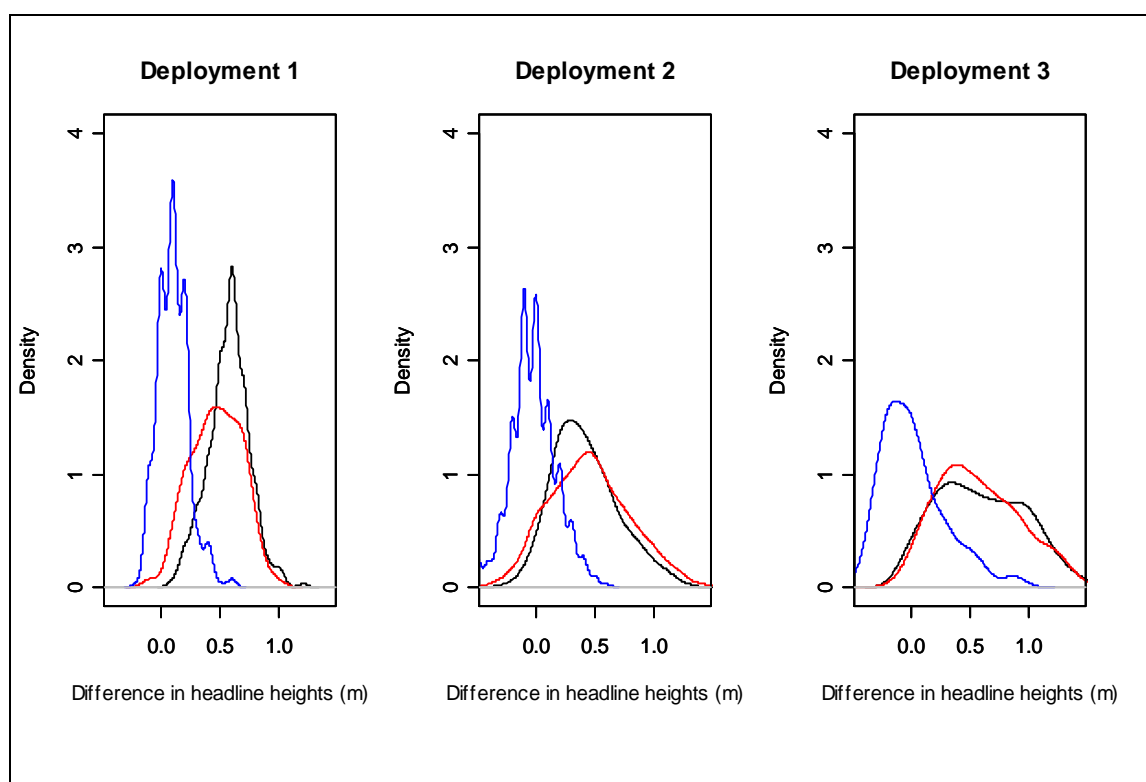
Figure 4: Density plots in the difference in float line heights between pairs of nets measured at 10 minute intervals. Blue line indicates difference between Standard and Stiffened net, Red line between Stiffened and $BaSO_4$ net, and Black line between Standard and $BaSO_4$ net.

The distribution of these differences can be explained by the changes in the heights of each nets float line relative to each other net during the length of each deployment. Fish catches fluctuated between deployments but were consistent for all nets within a

deployment. During the first deployment each net caught approximately one box of fish, during the second each caught three boxes of fish, and during the third deployment catches were low at approximately one quarter of a box of fish for each net. The continued reduction in fishing height of all three nets during the second deployment likely reflects the effect of increasing fish catch on float line height. Figures 5 shows the calculated float line height of each net in each of the deployments during the second trial.
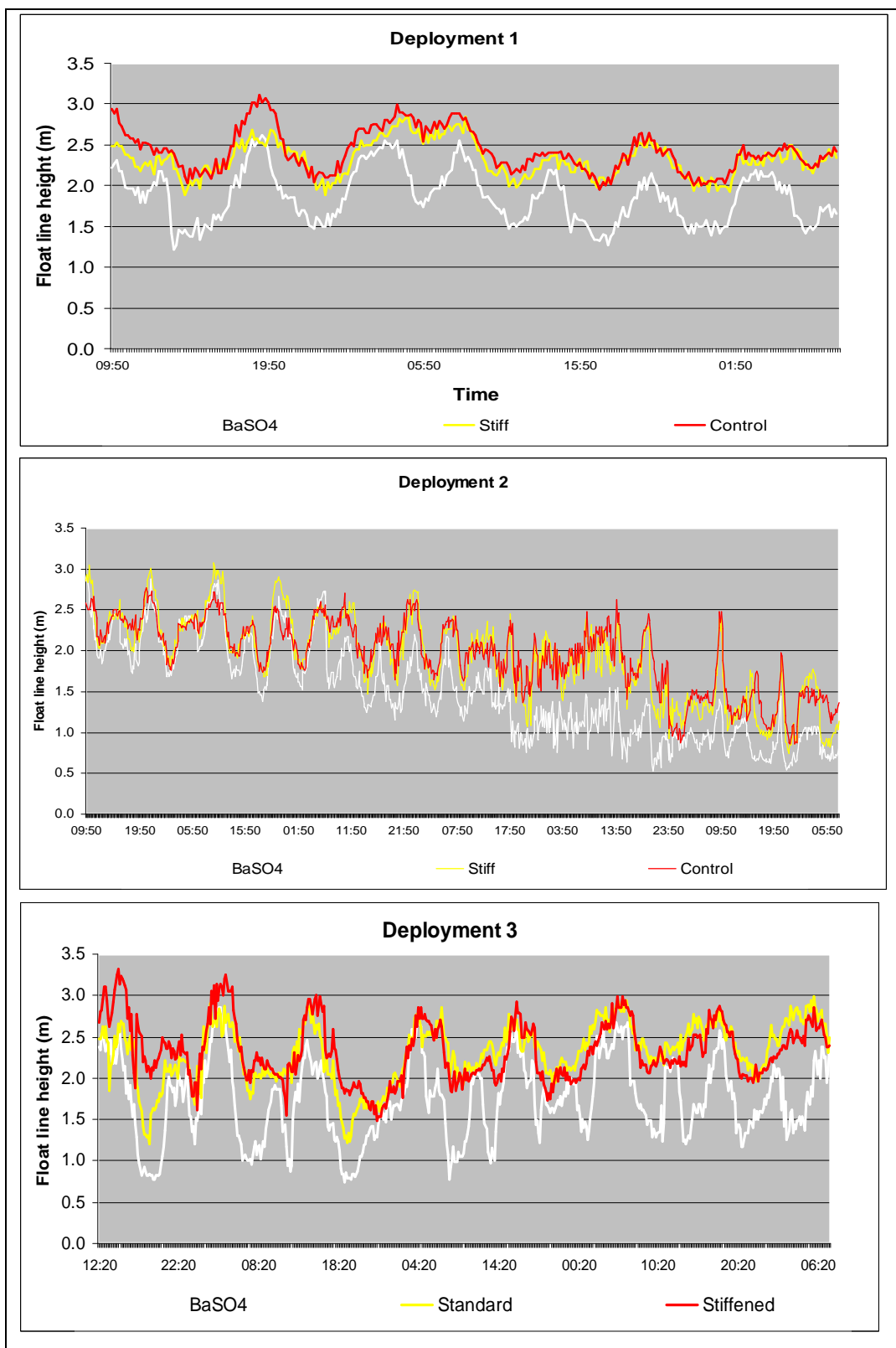
Figure 5: Calculated float line height of the three nets during each deployment. Time scales are not equal for deployments.

Tidal data were obtained from the Argentinean Naval Hydrographic website (www.hidro.gov.ar/Oceanografia) for the period of net deployment, only data on high and low water were available. Figure 6 shows high and low water heights and the corresponding net heights for those periods. In general, all three nets have highest float line heights when tidal heights were largest and lower headline heights during low water. Tidal range was greatest during the second deployment.

Figure 6: Float line height of three nets at low and high tidal ranges over the deployment period.
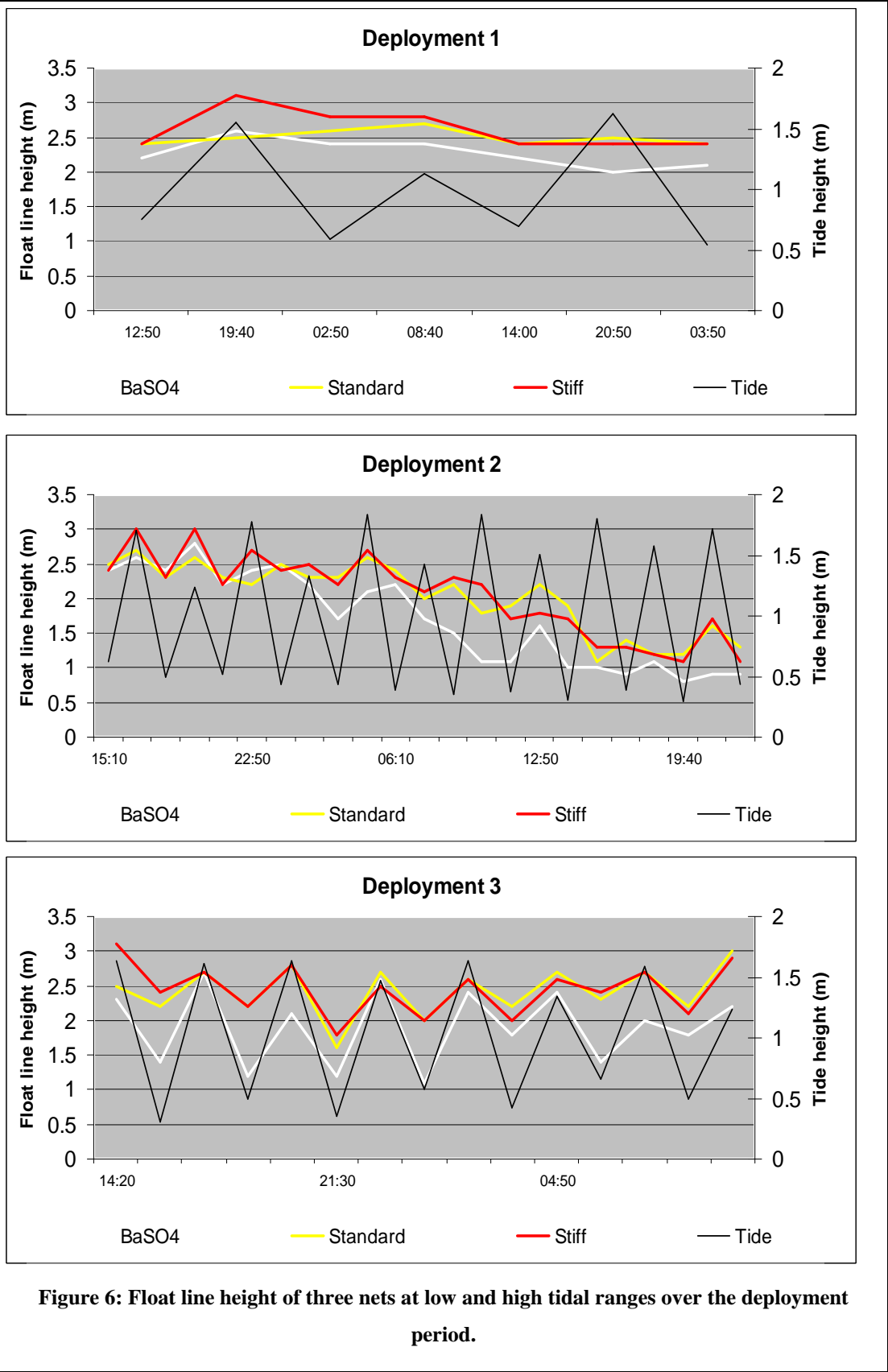
4.4.3 Analysis of observer data

A total of 283 hauls were observed between the 24th of October and the 11th of December 2009, during which 30 dolphins were bycaught in 28 hauls. Nets were set in three distinct areas (Figure 7).
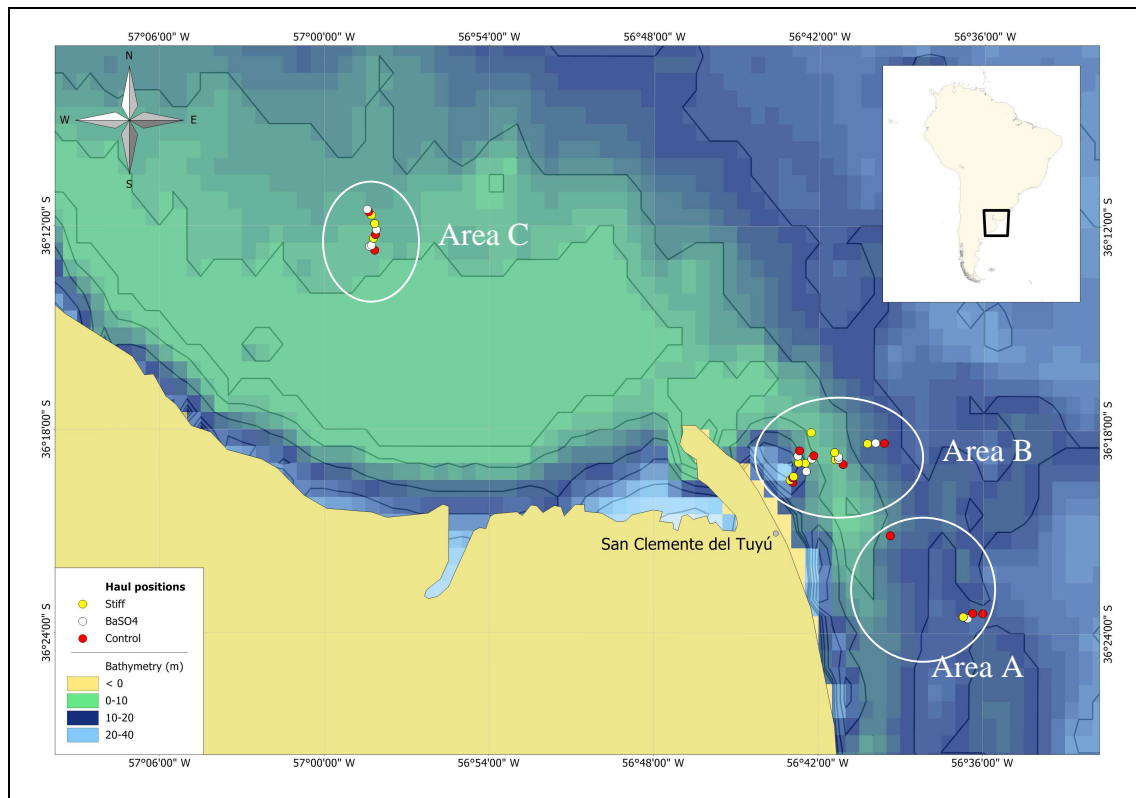
Figure 7: Location of observed hauls by coloured by net type.

The majority of nets in Area A were deployed approximately 8.5km from the shore in depths of 16-18m. Nets in Area B were deployed at distances between 2 and 7.5 km from shore in depths of 7 to12m. Nets in Area C were deployed approximately 13 km from the coast in depths of 4-5m. Observed bycatch rates at the end of this period were lowest in the Stiff net (Table 8). Two dolphins were caught in the same haul in a stiff net, and two dolphins were caught in the same haul in a control net. All other entanglements were of single animals.

Net	No. of hauls	Total soak durations (hrs)	No. of dolphins	Bycatch rate per haul	Bycatch rate per hour
Control	99	4415	14	0.14	0.0032
Stiff	98	4171	5	0.05	0.0012
BaSO$_4$	86	3607	11	0.13	0.0030

<div align="center">**Table 8: Summary of dolphin bycatch rates per net treatment.**</div>

GAMs were run as a way to inspect the relationship between dolphin bycatch rates and each explanatory variable. All candidate explanatory variables showed linear relationships with dolphin bycatch rates. The explanatory variable, vessel, was collinear with both latitude and longitude (GVIF>10)(See Chapter 2), which reflects the fact that each vessel generally fished in distinct areas. The final covariates offered to the model and associated GVIF scores are listed in Table 9.

Explanatory variable	GVIF	Df
Net position	1.057811	1
Net type	1.021558	2
Fish catch (kg per hour)	1.018216	1
Latitude	9.997045	1
Longitude	9.881751	1

<div align="center">**Table 9: GVIF for modelled covariates.**</div>

The only explanatory variables retained after step-wise model selection based on AIC were net type and longitude; however longitude was the only significant explanatory variable (Table 10).

Covariate	Parameter estimate	95% C.I.	p-value
(Intercept)	258.7	-1.2 to 518.5	P<0.05
Longitude	4.7	0.1 to 9.2	P<0.05
BaSO$_4$ net	-0.1	-0.9 to 0.7	p>0.1
Stiff net	-0.9	-2 to 0.1	p>0.1

<div align="center">**Table 10: Summary of parameter estimates, confidence intervals and significance levels for all covariates retained by the best mode.**</div>

Highest observed bycatch rates occurred in Area A, although effort here was only 44% of that observed in Area B (Table 11).

Area	Total soak duration (hrs)	No. of dolphins	Bycatch rate per hour.
A	3063	11	0.004
B	6922	17	0.002
C	2208	2	0.001

Table 11: Summary of bycatch rates per hour in the three fishing areas.

However, differences in bycatch rates relative to the amount of effort observed in each net type in each area were less (Table 12).

Area	Net	Soak time (hrs)	No. of dolphins	Bycatch rate
A	Stiff	1050	3	0.003
A	$BaSO_4$	1052	4	0.004
A	Control	962	4	0.004
B	Stiff	2305	2	0.001
B	$BaSO_4$	1977	6	0.003
B	Control	2641	9	0.003
C	Stiff	817	0	0.000
C	$BaSO_4$	578	1	0.002
C	Control	813	1	0.001

Table 12: Bycatch rates by net type in each area.

Because the anchors of nets were not brought onboard during hauling, in general, nets remained in the same location for a number of hauls. It is therefore possible that the positioning of nets relative to each other may introduce bias in bycatch rates if a certain type of net was always upstream or downstream of the other nets. While the data are too limited to allow a quantitative assessment of this, a general overview of of bycatch rates per net position can be obtained by calculating the bycatch rate for each individual for each location it was deployed (Figure 8).

Figure 8: Bycatch rates per net by average deployment location.
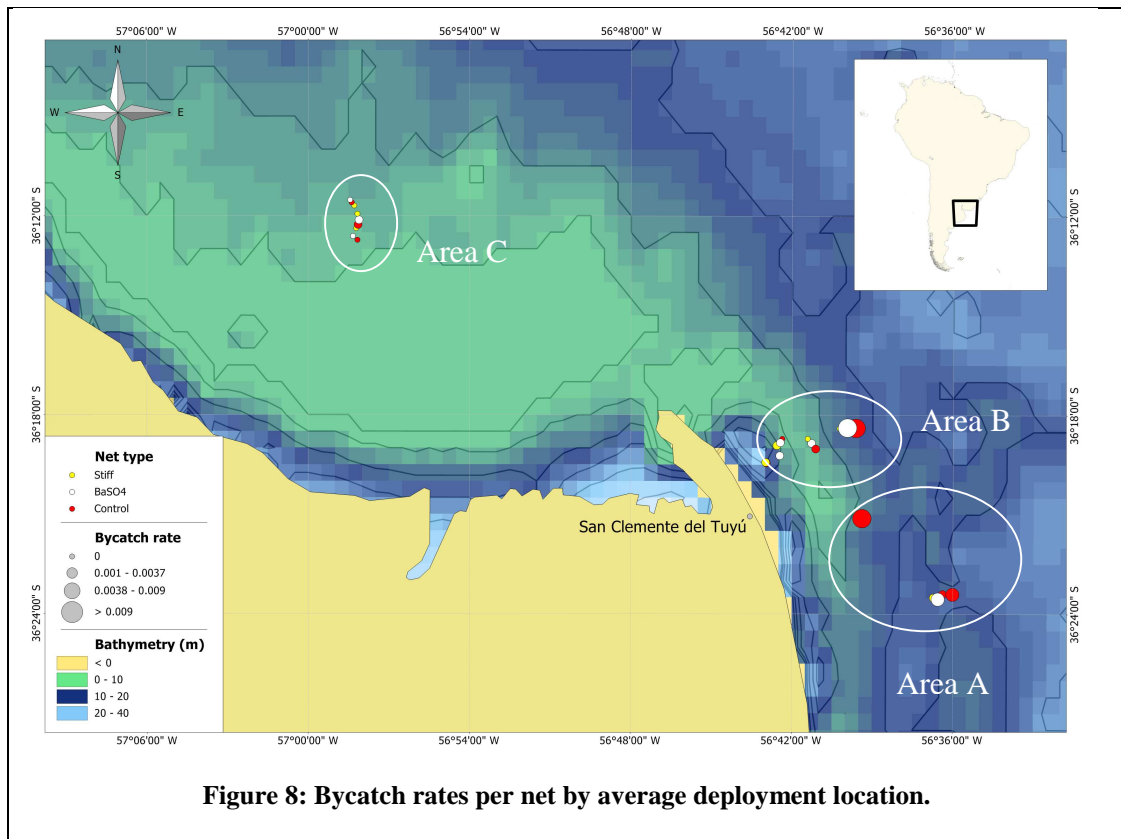
Bycatch rates per unit effort were generally highest in Area A. However, two net locations in Area B also had high bycatch rates, but these are a result of both nets catching a single animal during a soak time of 48 hrs. Figure 9 shows bycatch rates per net location in Area A. For the area where the three net types were deployed close together, lowest bycatch rates were recorded in the most inshore net.

Figure 9: Bycatch rates per net by average deployment location Area A.
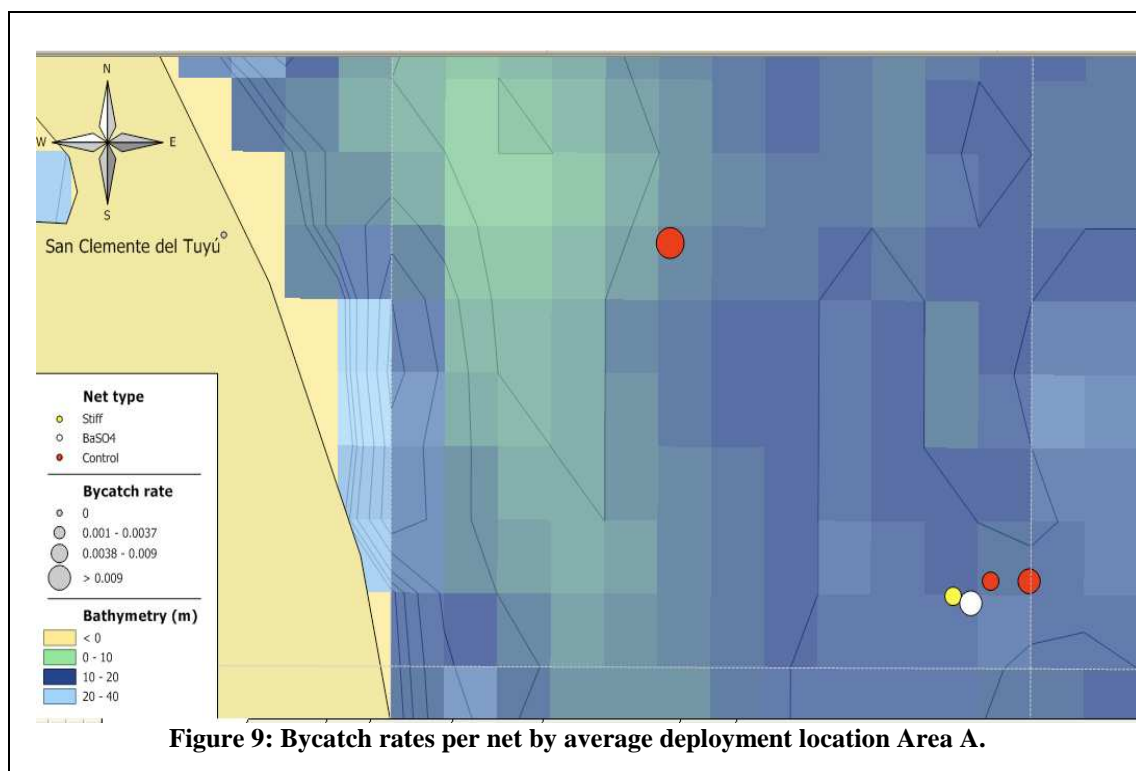
In Area B (Figure 10) highest bycatch rates occurred in nets positioned most southerly and those furthest offshore, while there was no clear effect of net position on bycatch rates in Area C (Figure 11). A more quantitative analysis of the effect of net position on bycatch rates may be possible when the data from the completed trial are available.

Figure 10: Bycatch rates per net by average deployment location Area A.

Figure 11: Bycatch rates per net by average deployment location Area A.
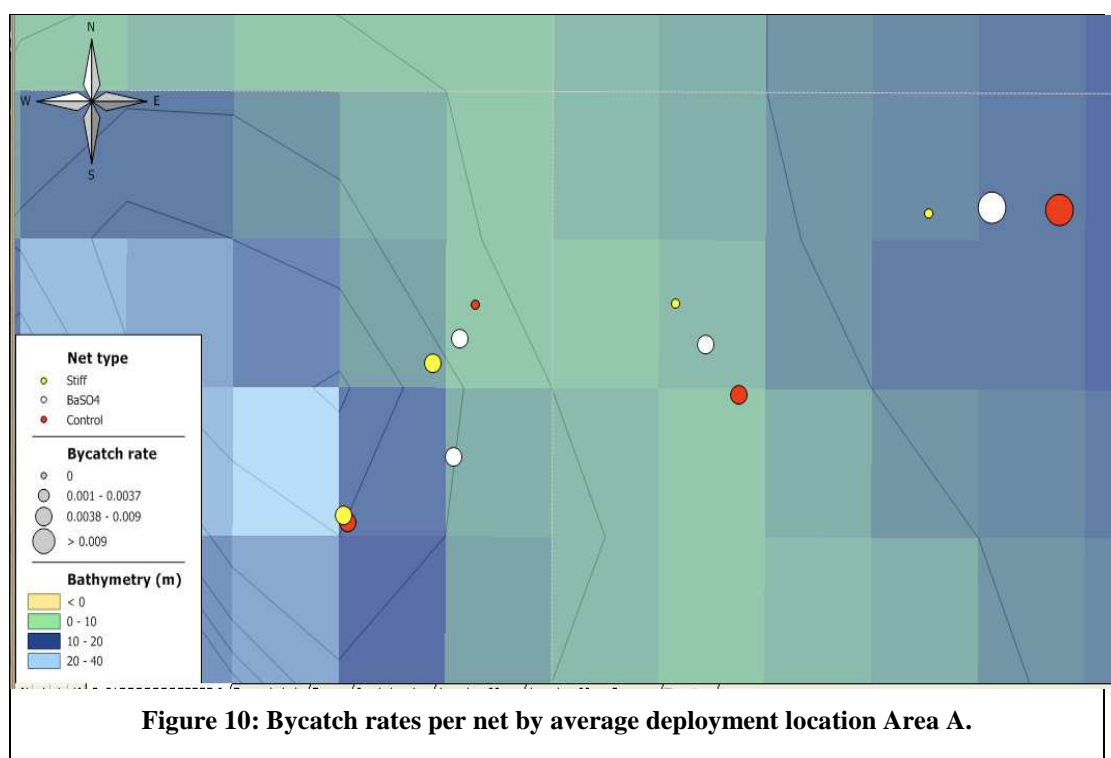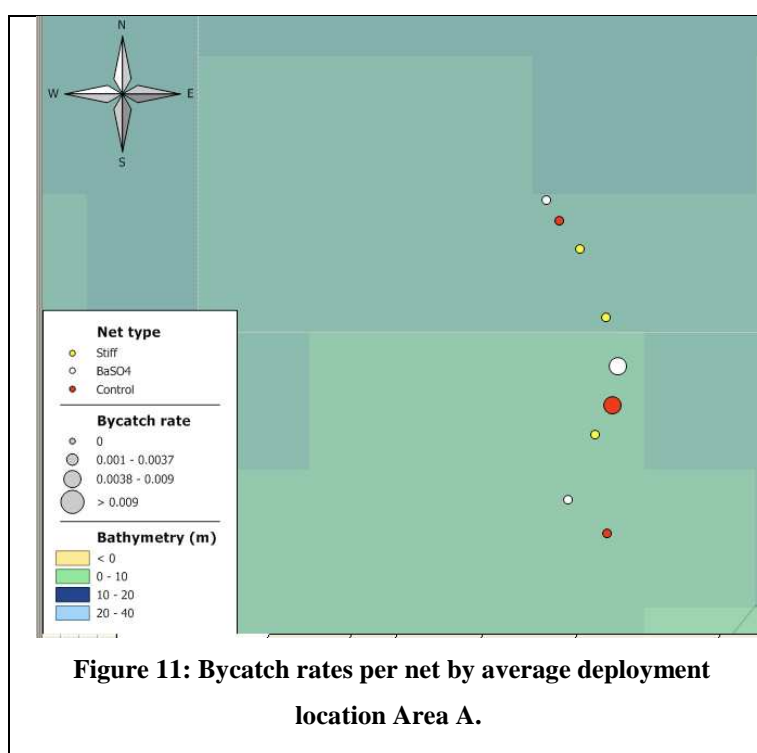
There were no significant differences in the total weight of fish caught between the BaSO$_4$ and control net ($p > 0.05$), or between the Stiff and control net ($p > 0.05$). Table 13 summarises the mean fish catch in kg per hour for each net type.

Net	Mean fish catch (kg/hr)	95% C.I.
Control	1.02	0.87-1.17
BaSO$_4$	0.93	0.77-1.09
Stiff	1.11	0.96-1.26

Table 13: Mean fish CPUE per net type

4.5 Discussion

Results of the second field trial showed that the mean float line height of the $BaSO_4$ net was significantly lower than the mean float line height of the control or Stiff net (p<0.001) for all deployments combined. In the first and third deployments both the control and Stiff net fished 70% of the rigged net area, whist the $BaSO_4$ net fished 50%. During the second deployment, the Stiff net fished 60% of the rigged net area, the Control net 50% and the $BaSO_4$ net 40%. The three nets remained in the same positions during each deployment and were set perpendicular to the current. Due to time constraints it was not possible to measure the float line heights of these nets when set parallel to the current. However, a study on the effect of current speed and orientation on net profiles found a greater reduction in float line heights for nets set perpendicular to the current (Stewart 1988). Therefore, the observed reduction in the float line height of the $BaSO_4$ net relative to the control net may not be as great if nets are set parallel to the current. Data collected by depth sensors during the first trial could not be used for quantitative analysis due to errors in depth readings by a number of the sensors, but results support the notion that relative differences between the three nets used in the trial are less when nets are set parallel to the current (Appendix 1). These data also show the effect of a reduction in fishing profile of two nets due to compression of plastic floats on the float line of the $BaSO_4$ and control net (Appendix 1 – deployment 2). Additional flotation was then added to these nets by fishermen and data collected during a second deployment of depth sensors on these nets indicate that the overall float line height of these nets was not increased (Appendix 1 – deployment 5).

No significant difference in the bycatch rates of Franciscana dolphins was found between the three nets during the trial. Although the number of observed hauls used for this analysis are lower than the number required to have sufficient power to detect a significant reduction in bycatch rates between the three treatments, observed bycatch rates in subsequent hauls also showed no significant difference in bycatch rates (P. Bordino, *pers. comm.).* These data were unavailable at the time of analysis. Multiple entanglements of two individuals occurred in one haul of a control net and one haul of a Stiff net; all other entanglements involved a single animal.

The only explanatory variable retained in the model was latitude and while net position did not appear to be a strong factor in observed bycatch rates, this was not quantifiable with the available data. Self-contained automatic click detectors (CPODs) were deployed with nets on three separate occasions during the first trial in order to assess the occurrence of Franciscana in the vicinity of the three net types, and to determine how the probability of bycatch was related to the presence of animals. However, all CPODs failed to record. Therefore, there is no information on the encounter rates for Franciscana in the three areas or the whether Franciscana are in the vicinity of gillnets more often than they are bycaught. Highest bycatch rates were observed in Area A where water depths were between 15 and 17 metres. Satellite tagged Franciscana dolphins generally swim at a depth less than 15m with deepest dives of 30 to 35m that correspond to the deepest depths in the area they were tagged (P.Bordino *pers. comm.*). Current speeds were also high in this area and nets were more susceptible to swell action than those deployed in the inner bay. Local fishermen believe bycatch rates are higher after stormy weather irrespective of soak time, while there is no quantitative data to support this view it is possible that increased water turbidity reduces the detectability of nets to echolocation clicks. There was no significant difference in the total catch of fish species per unit effort between the three net types. An anlaysis of fish CPUE and the distribution of fish lengths per target species should be conducted once the final data set is available.

In 2004, the IWC Scientific Committee supported the delineation of four distinct Franciscana Management Areas (FMAs) based on genetic evidence showing three genetically distinct populations within the range of this species (Secchi et al. 2003). FMA IV includes the area of San Clemente de Tuyu and Bahia Somborombon where data were collected in this study. However, analysis of additional genetic data from bycaught Franciscana suggests that a finer scale subdivision of FMA IV may be warranted (Mendez et al. 2008). Results of an aerial survey conducted in 2003 and 2004 estimated a density of 0.086 dolphins per km^2 for the northern section of FMA IV (Crespo et al. 2010), although the survey did not include Bahia Samborombon (Area C in this study). Crespo et al. (2010) used abundance estimates generated from the aerial survey to estimate bycatch removal rates in population terms for this species. Based on a bycatch mortality of 500 animals per annum in Argentine waters

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(Secchi et al. 2003) they calculated a removal rate from the population of 3-4%. Using higher bycatch estimates of 800 animals per year the bycatch removal rate from this population could be 5.6% or as high as 9.7% per annum. These calculated removal rates far exceed the estimated population growth rate of 2% per annum for this species (Secchi et al. 2003). Therefore, current estimates of bycatch rates are of serious conservation concern for this species and a significant reduction of mortality rates in bottom set gillnet fisheries is required for this population to persist.

There were no significant differences in observed bycatch rates of Franciscana dolphins in $BaSO_4$ nets or Stiff nets relative to control nets. While reflective nets have previously been shown to reduce significantly the bycatch rates of harbour porpoises (Trippel et al. 2003, Larsen et al. 2007, Trippel et al. 2009) it is possible that the lack of an equivalent reduction of Franciscana bycatch in these nets is related to the morphology of these species. Franciscana dolphins have elongated slim rostrums with up to 58 teeth in the upper jaw and 56 teeth in the lower jaw. Dolphins that were bycaught during this study frequently had their rostrums fully wrapped in the net with gillnet meshes caught between their teeth (Fig 12).

Figure 12: Two separate incidental captures of Franciscana dolphins, showing net material wrapped around the individuals' rostrums.

It is possible that this morphology means that if a Franciscana encounters a gillnet it has a high probability of becoming entangled regardless of the net material. Studies

investigating the mechanism by which captive harbour porpoises became entangled in gillnets found that entanglements occurred when the head, flipper, fluke or dorsal fin came into contact with the net (Kastelein et al. 1995). The presence of tubercules on the dorsal fin ridge and tail fluke of harbour porpoises might cause gillnet twine to be caught on them preventing the net from sliding off the fin. Larsen et al. (2007) postulated that the reduction of harbour porpoise bycatch in Iron Oxide nets may have been due to the meshes being less susceptible to catching these tubercules as a result of the additional stiffness of the net material.

Given the results of the field trial in Argentina, the use of Stiff or acoustically reflective nets does not seem to hold much promise in reducing Franciscana bycatch rates in bottom set gillnet fisheries. A trial by Bordino et al. (2002) showed that pingers could significantly reduce bycatch rates of this species; however, the presence of pingers also led to a significant increase in net depredation by sea lions (*Otaria flavescens*). An investigation into catch rates of target and non-target species using hand lines instead of gillnets found that while hand lines caught the same species as gillnets the CPUE was lower. However, bycatch rates of non-target species were significantly reduced (P.Bordino *pers comm*.). Hand lines may therefore provide a suitable alternative gear to reduce the fishing effort in the bottom set gillnet fishery in this region.

The uptake of this fishing method by fishermen is dependent on a number of factors. Although hand lines are cheaper than gillnets, more effort is required to catch the same biomass of fish as caught in gillnets. However, the quality of the target species was higher when caught by hand lines. Therefore, if a market for this higher quality fish existed then the economic benefits of using hand lines may outweigh the additional labour required to fish this gear. Other management strategies suggested to reduce the bycatch of Franciscana include moving effort into deep waters where the species is less abundant or banning fishing in nursery areas for Sciaenid fish (Crespo et al. 2010), the main prey species of these dolphin.

4.6   References

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4.7 Appendix 1

The following graphs show the fishing behaviour of the standard and two modified nets collected during the first experimental trial using depth loggers.

Deployment 1: 26/10/2009-29/10/2009 Area B

The three experimental nets were set in approximately 7m of water and were hauled once during the 4 days that the sensors were deployed. The largest tidal range during this period was 0.6m – 1.35m. The nets were set parallel to the current. Figures A1 and A2 shows the active fishing heights of each net during the first deployment.

Figure A1: Active fishing heights of the three nets during the first haul.
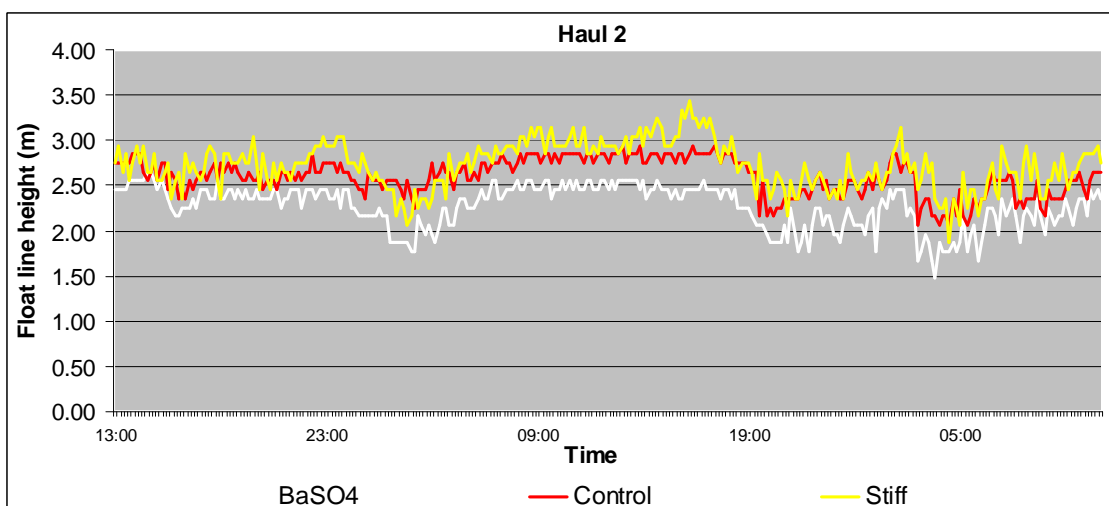
Figure A2: Active fishing heights of the three nets during the second haul.

Deployment 2: 01/11/2009 – 05/11/2009 Area A

Nets were set in approximately 15m of water and were hauled three times during the deployment period. The largest tidal range during this period was 0.38m – 1.37m. All three nets were set parallel to the tide. Figures A3a-A3c show the active fishing heights of each net. The fishermen added additional floats to both the red and white net after the second haul to compensate for the deformed floats on the float lines of each of these nets.

Figure A3a: Active fishing heights of the three nets during haul 1.

Figure A3b: Active fishing heights of the three nets during haul 2.

Figure A3c: Active fishing heights of the three nets during haul 3.

Deployment 3: 10/11/2009 – 12/11/2009 Area B

The three experimental nets were set in approximately 7m of water in the same location as the nets during the first deployment, but perpendicular to the current. The largest tidal range during this period was 0.37m – 1.65m. Figures A4 shows the active fishing heights of each net.
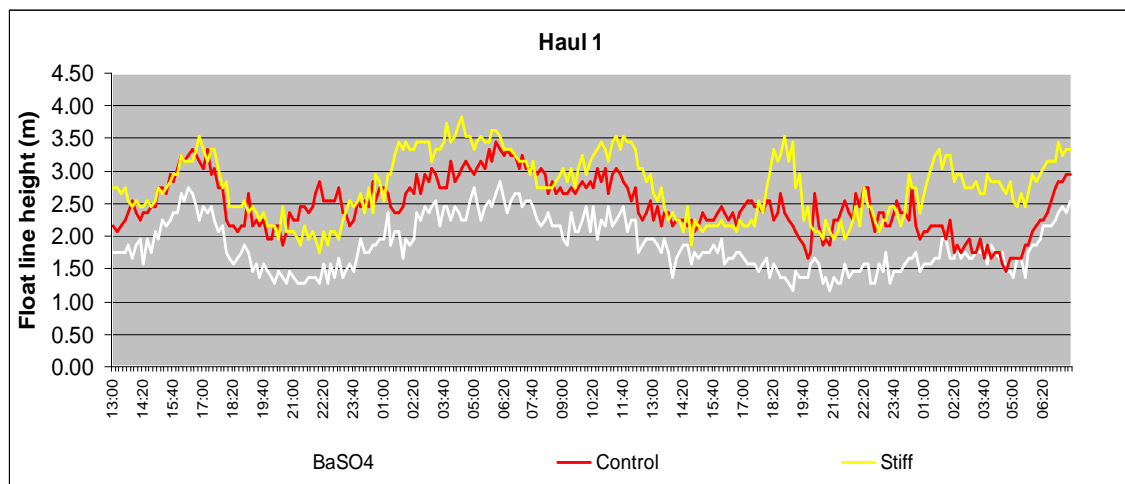
Figure A4: Active fishing heights of the three nets during the third deployment.

Deployment 4: 15/11/2009 – 20/11/2009 Area B

The three experimental nets were shot in 6m of water and were set perpendicular to the flow of the current. They were also fished without bridles. The largest tidal range

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during this period was 0.36m – 1. 5m. Figures A5a & A5b show the active fishing heights of each net during this deployment.
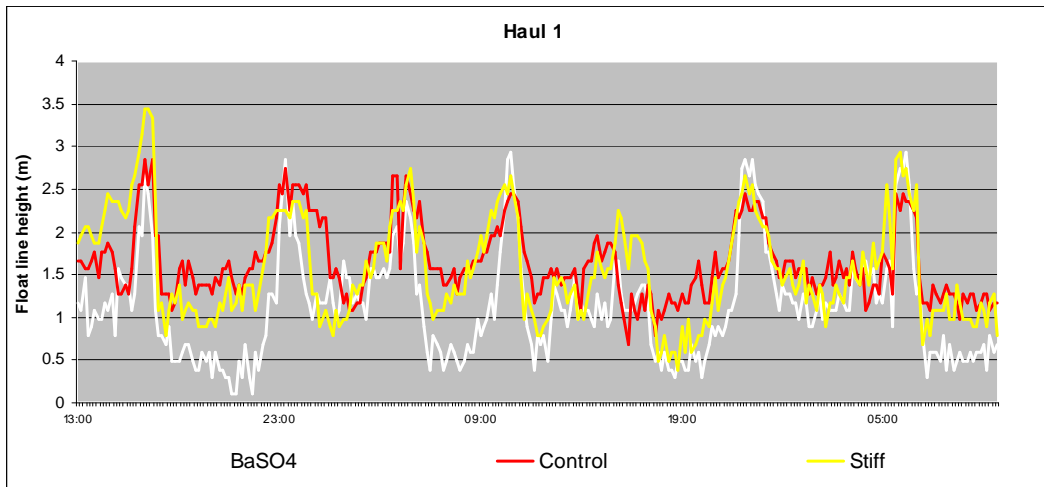
Figure A5a: Active fishing heights of the three nets during the first haul. Nets were fished without bridles.
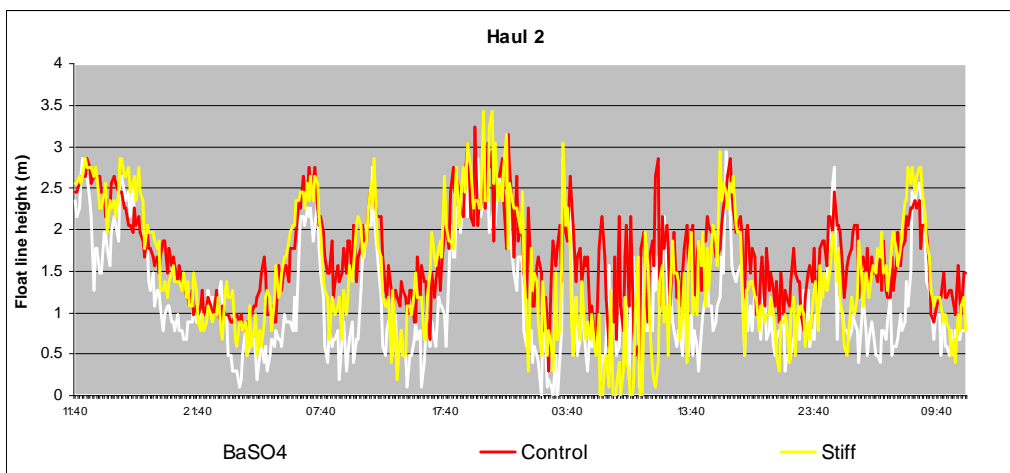
Figure A5b: Active fishing heights of the three nets during the second haul. Nets were fished without bridles.

During the second haul a dolphin was incidentally captured in the Stiff net. It is likely that the sudden decrease in depths recorded by the sensors at approximately 22:40 may indicate the capture event and the dolphin then pulling the net towards the surface in an attempt to escape (Figure A6).

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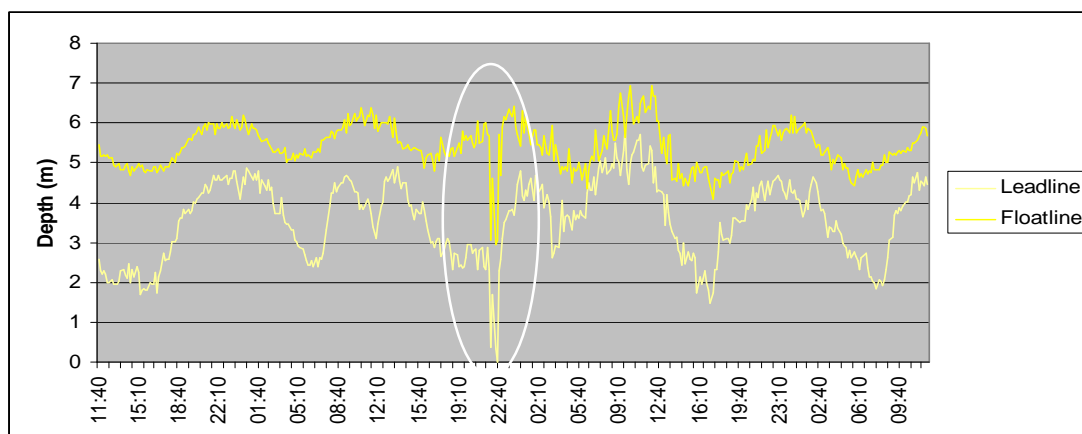
Figure A6: Headline and float line of stiff net during haul 2.

Deployment 5: 22/11/2009 – 26/11/2009 Area B

The three experimental nets were shot in 8m of water and were set parallel to the flow of the current. They were hauled twice during the deployment period and the greatest tidal range was 0.51-1.46m. Figures A7a & A7b show the active fishing heights of each net during this deployment. These were the same nets used in deployment 2 and the low fishing height of the control and barium sulphate nets reflect the problems with the flotation of these nets. The control net appears not to have fished properly during the second haul, and the total fish caught (kg) in this net was only 16% of the catch in the stiff net.
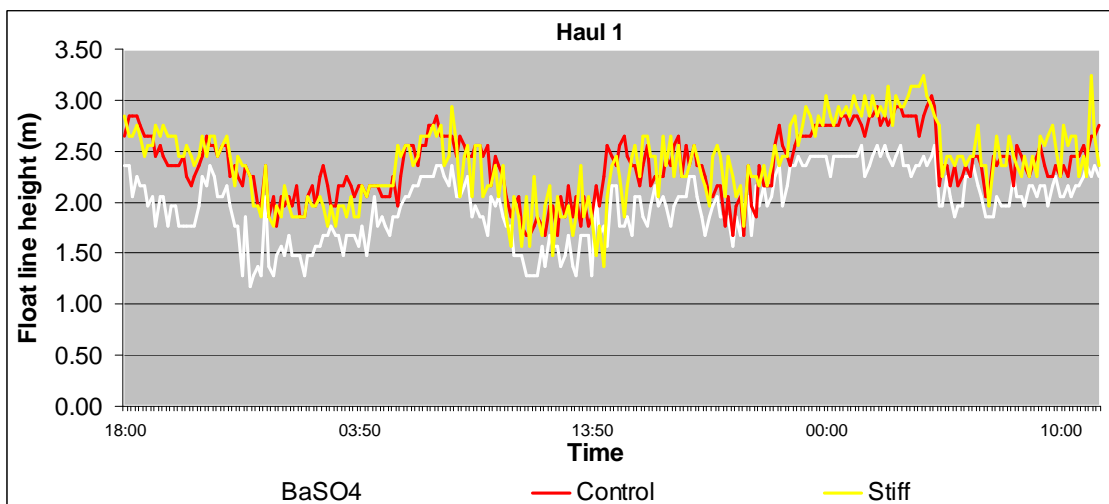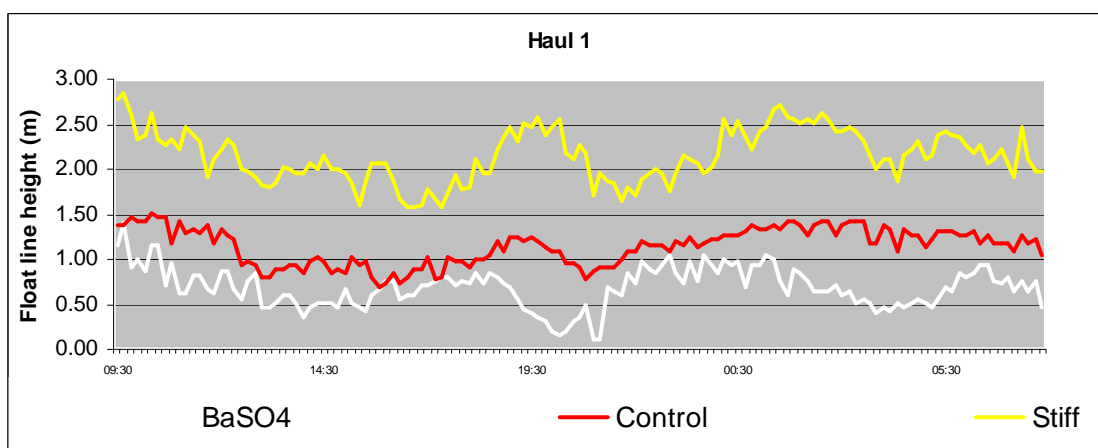

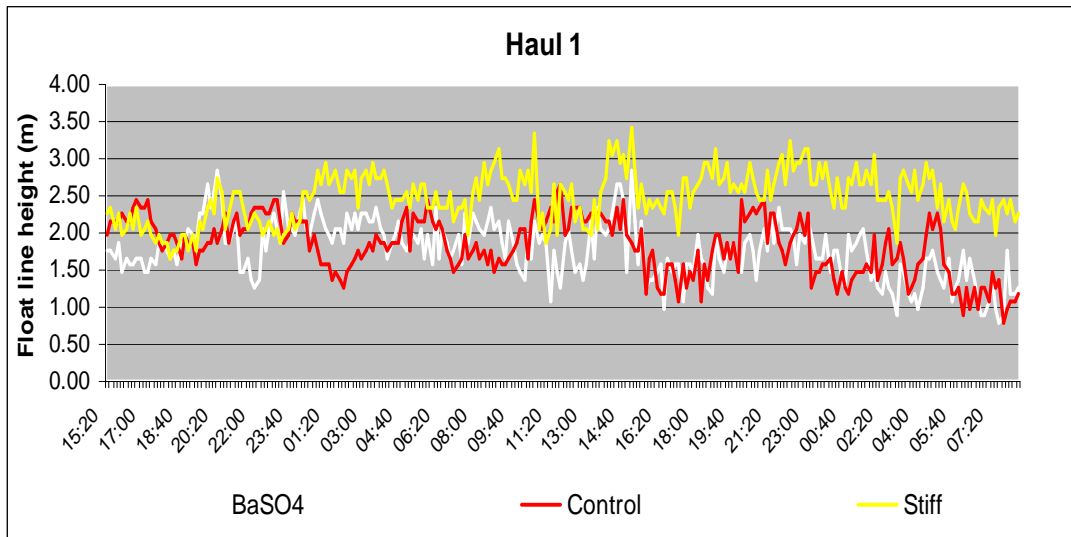**Figure A7a: Active fishing heights of the three nets during the first haul.**
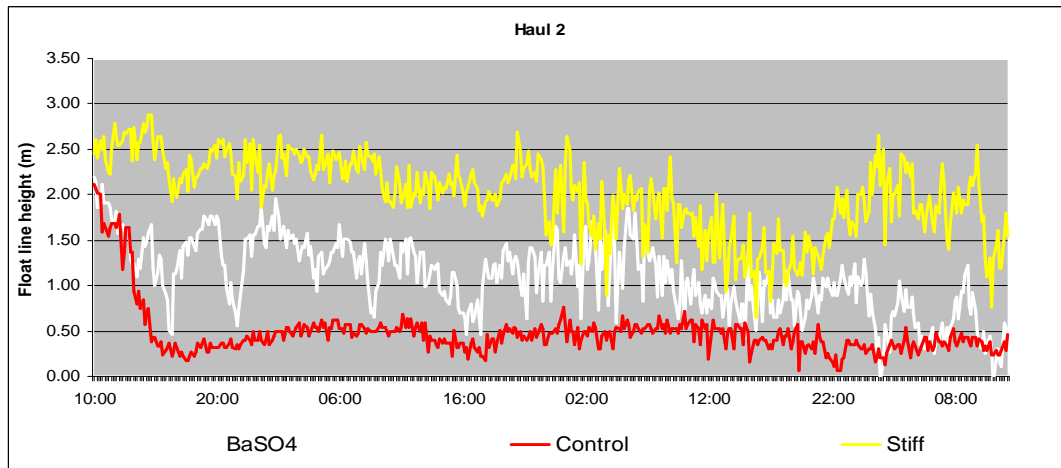
Figure A7b: Active fishing heights of the three nets during the secomd haul.

Deployment 6: 29/11/2009 – 05/12/2009 Area C

The three experimental nets were shot in 6m of water and were set parallel to the flow of the current. They were hauled twice during the deployment period and the greatest tidal range was 0.51-1.46m. Figures A8a-d show the active fishing heights of each net during this deployment. Three replacement tags were used in this deployment, but were not calibrated in the pool at Mundo Marina. Therefore, only offset values from the manufacturer were available. These offsets were adjusted based on the observation in all other deployments that the Stiff and control net fished at relatively equal heights, so it is possible that the errors associated with predicted float line heights are greater than in previous deployments.
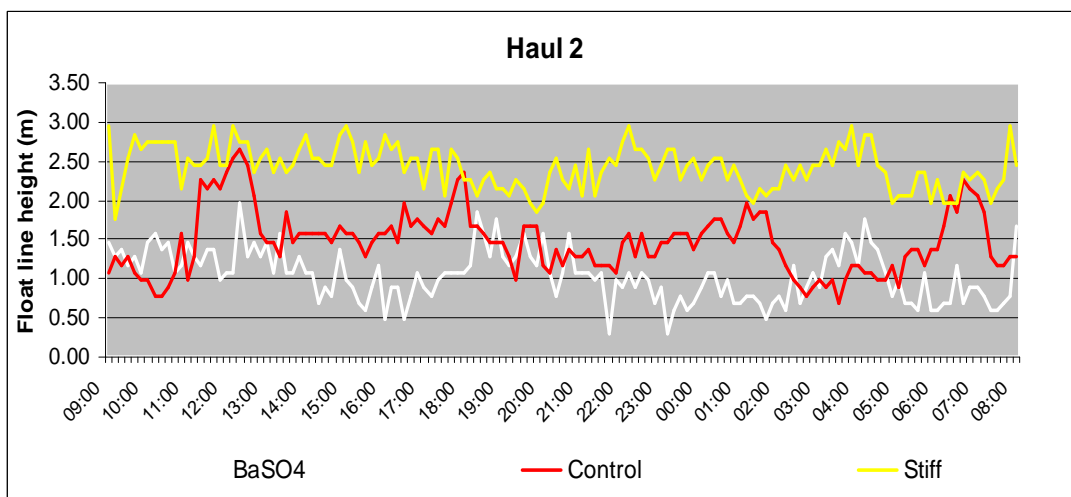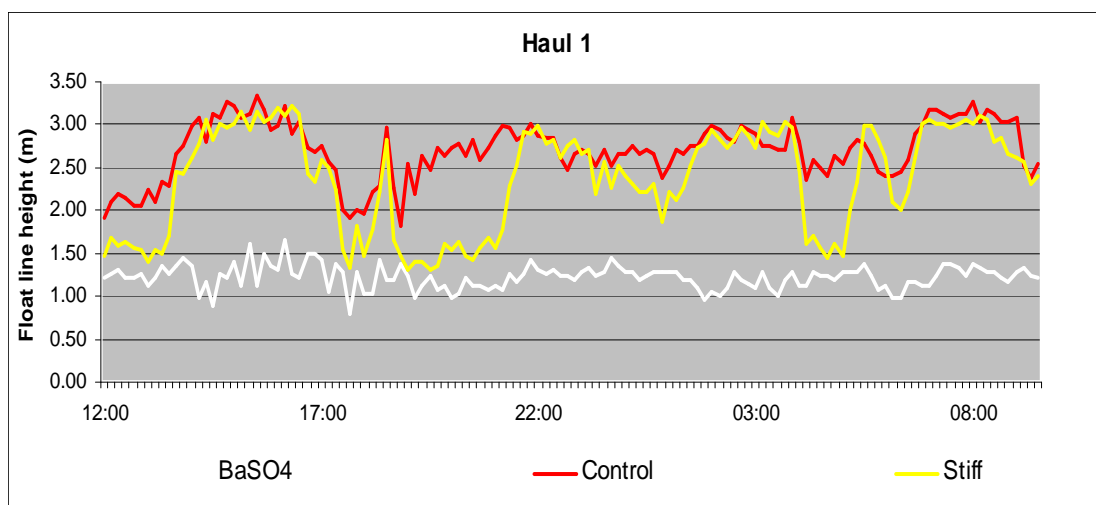

Figure A8a: Active fishing heights of the three nets during the first haul.

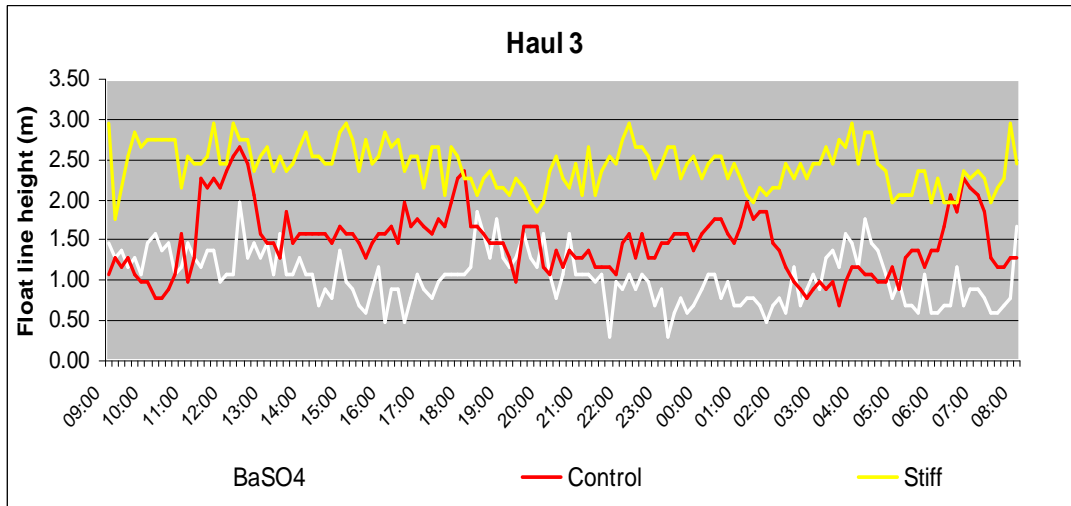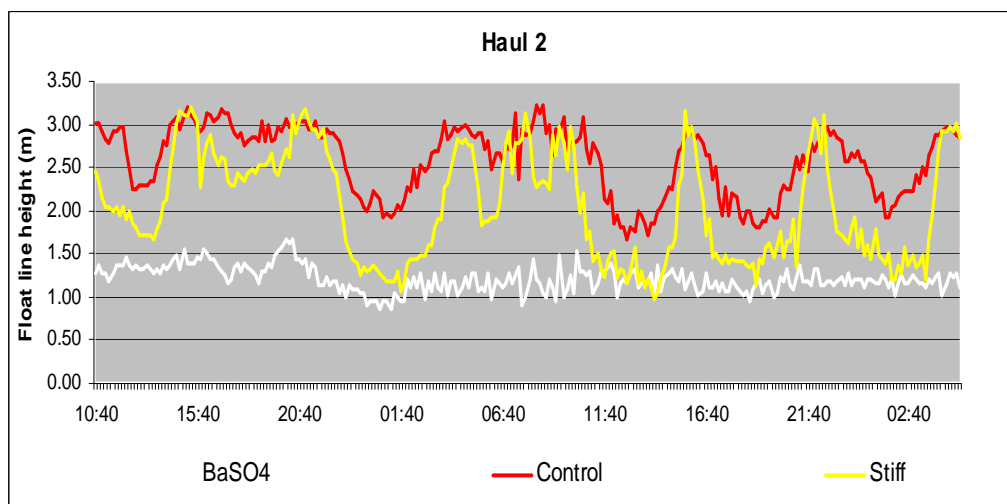**Figure A8b: Active fishing heights of the three nets during the second haul.**
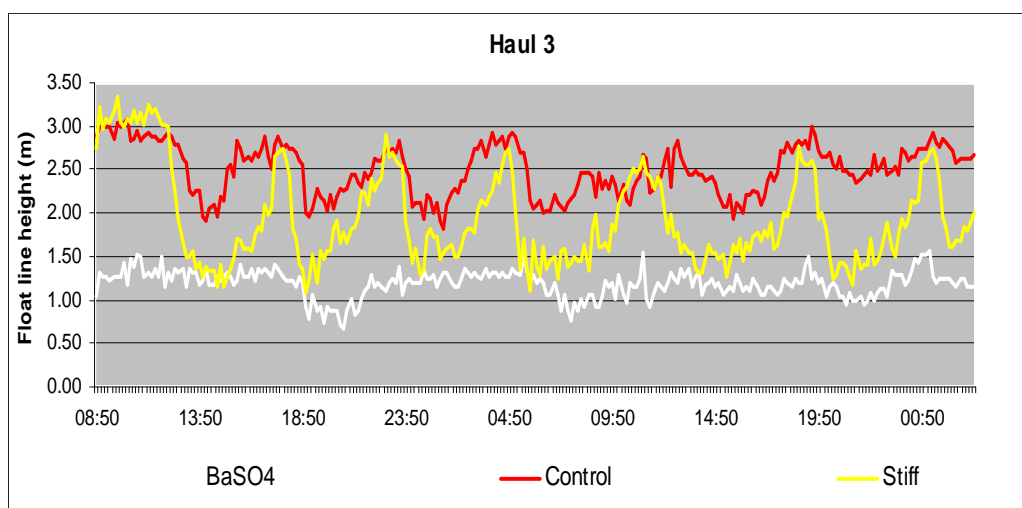
Figure A8c: Active fishing heights of the three nets during the third haul.
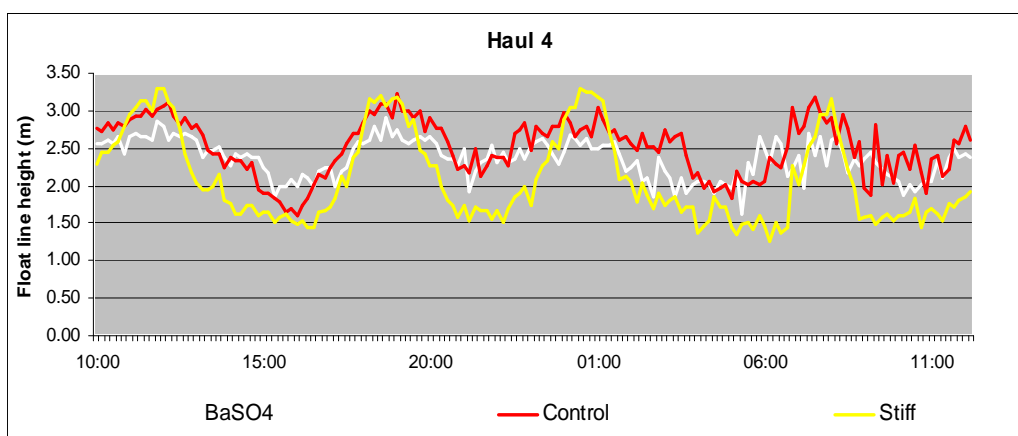
Figure A8d: Active fishing heights of the three nets during the four haul.

During the first three sets, the float line of the BaSO$_4$ net did not appear to fish properly, however there was no indication that this net was twisted when it was hauled.

During the third haul, a dolphin was bycaught in the control net. Figure A9 shows the depths of the float line and lead line of this net. There is no apparent sudden reduction in float line or lead line height during this haul, as was noted in the second haul from deployment 4 where bycatch also occurred. As sensors were set to measure at 10 minute intervals it is most likely that the dolphin became entangled during a period when depth measurements were not being collected.
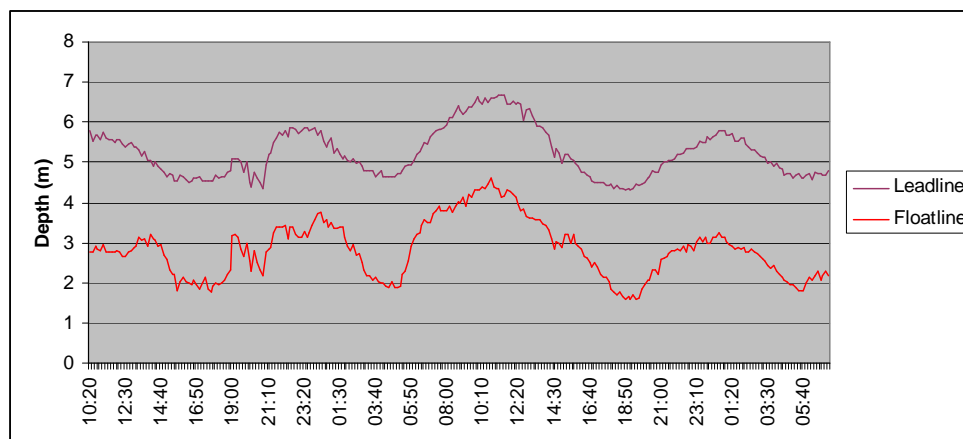
Figure A9: Headline and float line of the control net during haul 3.

5. Chapter 5: Investigating the echolocation behaviour of harbour porpoises in the presence and absence of bottom set gillnets.

5.1 Abstract

Harbour porpoises (*Phocoena phocoena*) have been recorded in the presence of bottom set gillnets much more frequently than they are bycaught. However, a comparison of the occurrence and echolocation of harbour porpoises in the vicinity of gillnets to when no gillnet was deployed has not previously been examined. CPODs were used to record the echolocation behaviour of harbour porpoises in the presence or absence of bottoms set gillnets in an experimental trial in Bridlington Bay, North Yorkshire. There was no significant difference in the rate, length or intensity of encounters recorded by PODs deployed with or without a net. However, the proportion of fast click trains, which are linked to navigation and prey capture, were statistically higher when a net was present. Rising click rates in sequences of trains have been attributed to harbour porpoise adapting their bio sonar to account for the two-way travel time of an echolocation click as they approach a landmark or prey item. Rising click trains recorded on CPODs deployed both with and without nets indicates that the presence of CPODs affect the echolocation behaviour of harbour porpoises.

5.2 Introduction

Harbour porpoises are frequently bycaught in bottom set gillnets in areas where their ranges overlap with gillnet fisheries. Although the mechanism(s) by which entanglements occur are not well understood, several hypotheses exist. These include that harbour porpoise are unable to detect gillnets at sufficient distances to avoid them or else are not echolocating continuously; that they can detect the nets but do not perceive them as a barrier; or that individuals foraging in the vicinity of nets may get distracted by prey resulting in entanglement. One of the major reasons for this gap in understanding is the difficulty of collecting behavioural data of free ranging porpoises while they are in the vicinity of gillnets. One methodology that has been used to record harbour porpoise presence or behaviour around nets is passive acoustic monitoring (PAM) (SMRU et al. 2001, Cox & Read 2004, Koschinski et al. 2006),

and PAM monitoring is becoming a standard method for studying cetacean behaviour in general.

Harbour porpoise produce narrow band, high frequency echolocation clicks with peak frequencies between 115-145 kHz and a narrow transmission beam of 16° (Mohl & Andersen 1973, Goodson 1996, Au et al. 1999). To date, most studies on harbour porpoise echolocation behaviour have been conducted on captive porpoises. These studies have improved our understanding of echolocation behaviour during foraging and prey capture (DeRuiter et al. 2009, Verfuß et al. 2009) target detection (Kastelein et al. 1999, Atem et al. 2009) orientation (Verfuß et al. 2005) and socialising (Clausen et al 2010). Other studies have aimed to elucidate the distances at which porpoises can theoretically detect bottom set gillnets and the reactions of captive animals to nets in their enclosures (Au et al. 2007, Kastelein et al. 1995a, Kastelein et al. 1995b, Kastelein et al. 2000, Mooney et al. 2004).

In contrast, few studies have collected behavioural or acoustic data from free ranging animals. Satellite tags have provided information on the diving behaviour and movement patterns of harbour porpoises (Westgate et al. 1995, Teilmann et al. 2007), while the use of acoustic data loggers has provided an insight into the way that free ranging animals use their bio sonar. For example, Akamatsu et al. (2007) showed that a tagged free ranging harbour porpoise echolocated frequently, producing click trains on average every 12.3 seconds. When the porpoise was not echolocating, periods of silence were generally shorter than the calculated sonar range of the animal. 90% of the echolocation click trains recorded by the tag were produced within 20 seconds of the previous click train and the mean recorded inter click interval (ICI) was 80.5 ms. The authors note that these results are likely to be a conservative estimate of harbour porpoise echolocation abilities, as the specifications of the acoustic logger used may have resulted in clicks with a low source level not being recorded. Results from studies of both captive and free ranging animals, can inform our interpretation of data on harbour porpoise echolocation activity and behaviour collected by (PAM).

The use of self-contained automatic click detectors, such as TPOD and CPOD porpoise detectors (Chelonia Ltd., www.chelonia.co.uk), which can be deployed for long periods at sea, has increased in recent years. In general, these studies have

focused on assessing seasonal changes in harbour porpoise echolocation occurance (Verfuß et al. 2007), diel variation in echolocation use (Carlstrom 2005, Todd et al. 2009), behavioural reactions to noise and anthropogenic activities at sea (Cox et al. 2001, Culik et al. 2001, Koschinski et al. 2003, Carstensen et al. 2006, Carlstrom et al. 2009) and to assess the acoustic behaviour of harbour porpoise around standard and modified static gillnets (Cox & Read 2004, Koschinski et al. 2006). Two of these studies have further analysed TPOD data to infer foraging activity in harbour porpoises (Carlstrom 2005, Todd et. al 2009).

The inter click interval (ICI) represents the time it takes for an outgoing echolocation click to return as an echo, plus a lag time for signal processing of that echo. The inter-click-interval (ICI) has been shown to decrease as an animal locks its sonar onto an item during navigation (Verfuß et al. 2005) and very fast click trains have been recorded during prey capture (DeRuiter et al. 2009, Verfuss et al. 2009). Information on the ICI of harbour porpoise echolocation click trains can be collected by PAM. Two studies using such data, collected by TPODs, postulated that short ICIs could indicate that harbour porpoise were acoustically investigating their environments in greater detail or could represent foraging activity (Carlstrom 2005, Todd et al. 2009).

If short ICIs are indicative of either porpoises investigating objects in detail or of foraging behaviour, then these data could be used to assess the behaviour of harbour porpoises in the vicinity of gillnets. PAM can be used to collect such data, as self-contained click loggers can easily be deployed with gillnets, and can record for the duration that nets are fishing. Although it has been hypothesised that harbour porpoise may not be able to detect nets in time to avoid them, studies utilising passive acoustic monitoring have shown that harbour porpoise are in the vicinity of nets much more frequently than bycatch occurs (SMRU 2001, Cox & Read 2004). For example, SMRU et al. (2001) found that in a 24 hour deployment of a TPOD on a bottom set gillnet, approximately 40% of hours contained at least one harbour porpoise detection, while no bycatch occured.

In the same study SMRU et al. (2001) found that harbour porpoise bycatch rates were significantly higher in nets with a buoyant float line made of rope, with a polystyrene core, than in nets with a standard polypropylene float line and additional plastic floats.

The authors postulated that the buoyant float line rope may have changed the behaviour of the net while fishing, such as lowering or increasing the float line height, or may have been less conspicuous to echolocating harbour porpoise than the floats on the standard propylene headline. Results from Chapter 3 indicated that the cigar rigged float line net fished a lower proportion of relative net height than a net rigged with a continuous buoyant float line.

It has been hypothesised that harbour porpoises may be attracted to struggling fish caught in static gillnets (Gaskin 1984). However, as yet there is no evidence to suggest that porpoises are feeding around gillnets. One study, looking at the stomach contents of hake and bycaught harbour porpoises, showed no overlap in ingested prey (Kindt-Larsen 2007). SMRU et al. (2001) also found no clear relationship between the amount of fish in the net and the amount of echolocation click activity recorded.

Therefore there were three main objectives for this chapter. The first was to investigate the echolocation behaviour of harbour porpoises in the presence and absence of bottom set gillnets, to determine whether porpoise are "attracted" to nets. The second was to investigate whether echolocation behaviour varied with float line type. The final objective was to investigate whether data collected by PAM could be used to determine if harbour porpoises are foraging in the vicinity of gillnets.

5.3   Materials and Methods

Two separate trials were conducted in Bridlington Bay, North Yorkshire in 2007 and 2009. Fish catches in Bridlington have been declining over the last decade and a large proportion of commercial fishing boats in the region have switched from gillnet fishing to lobster pots. Fish catches in experimental nets in 2009 were very low, and as one of the objectives of this chapter was to investigate whether harbour porpoises forage around bottom set gillnets, it was likely that data collected in Bridlington Bay would not be truly representative of harbour porpoise interactions with gillnets in areas with higher fish catches. Therefore the Cornish Wildlife Trust (CWT), who had deployed a number of CPODs on gillnets in a commercial fishery during a pinger

trial, kindly provided data recorded on seven CPODs deployed with un-pingered net tiers, for further analysis.

5.3.1.1   2007 trial

A homogenous fishing ground consisting of a sandy benthos close to Bridlington Harbour, Yorkshire, was chosen to minimise the influence of habitat type on harbour porpoise occurrence. Five fishing locations at a spacing of 1km were selected for the experiment, with each location being assigned a letter code. The placement of nets was randomly determined, and a TPOD (Chelonia Ltd., www.chelonia.co.uk) was deployed in the middle of each net. TPODs are fully automated passive acoustic monitoring systems that log tonal clicks occurring in a frequency range of 20 to 150KHz. The time of occurrence, centre and duration of each click is logged and then processed using TPOD.exe computer software which assigns clicks to trains and then classifies these trains as being from cetaceans (porpoise or dolphin species), sonar or other sources. An additional TPOD was deployed in each of two fishing locations without a gillnet. Figure 1 shows an example of net and TPOD deployment for three successive trials.
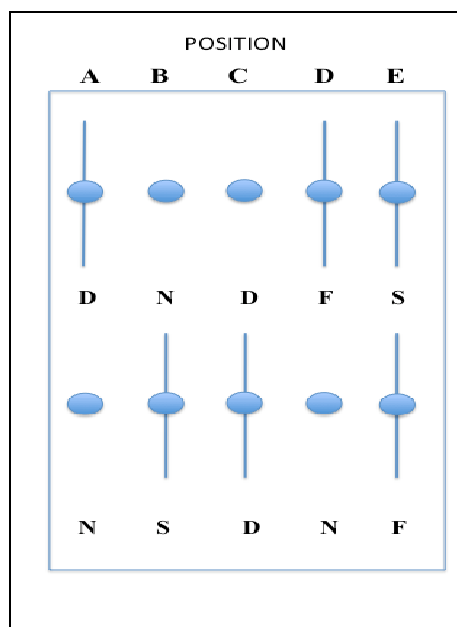
Figure 1: Schematic showing an example of gillnet deployments in three successive trials. D = net with double flotation, S=net with single flotation, F=net with floats, N= no net.

The nets deployed in the study were standard and modified turbot gillnets. The characteristics of each net are summarised in Table 1. Two tiers of nets were rigged with different amounts and types of flotation. The "Single" net was rigged with the standard amount of flotation consisting of a single 9.5mm float line and 3.6mm lead line. The "Double" net was rigged with a single 12mm float line and 9.4mm lead line and the "Floats" net was rigged with a nominal 10mm braided polypropylene float line with 6 inch polystyrene cigar floats spaced at 5m intervals. A short tier length of 200m was chosen to minimise the amount of net in the water to reduce the likelihood of porpoise bycatch.

Tier	Mesh size (inches)	Twine diam.	Height (in meshes)	Float line diameter (measured)	Lead line diameter (measured)	Length of net panel
Cigar floats	10	0.58	6 ½	9.5 mm	3.6 mm	100m
Double float line	10	0.58	6 ½	12 mm	9.4 mm	100m
Single float line	10	0.58	6 ½	10 mm	5 mm	100m

Table 1: Summary of gillnet characteristics.

The experimental nets and TPODs were first shot on the 24th of July 2007 and were rotated every 48 hours, with a total of 4 deployments and hauls. TPODs were downloaded on the 1st of August and initial data analysis showed an unexpectedly high variation in detections between TPODs. For this reason the experiment was halted to calibrate the PODs. All five TPODs were placed on a single anchor and deployed for 2 days. Four of the TPODs recorded for a total of 66 hours each whilst the fifth switched off after 4 hours.

Due to this wide variation in the number of detections positive minutes per hour recorded by the four TPODs during the calibration, the manufacturer was contacted. After consultation, a second calibration was conducted in St. Andrews, with modified detection settings for the TPODs. Table 2 shows the range in detection positive minutes (DPM) per hour during each calibration.

POD ID	Clicks per hour	
	1st Calibration	2nd Calibration
280	20.5	30.29
298	10.91	17.53
299	25.57	3.83
312	00.42	0
313	100.06	61.56

Table 2: Detection positive minutes per hour during first and second calibration.

After further consultation with the manufacturer it was apparent that the variation in POD sensitivities could not be fixed and the trial was halted.

5.3.1.2   2009 Trial: Study area

The second trial was conducted in Bridlington Bay, North Yorkshire between the 8th of July and 20th of August, 2009. Again, a homogenous fishing ground consisting of a sandy benthos was chosen to minimise the influence of habitat type on harbour porpoise occurrence. Eight Chelonia CPOD V0 porpoise click detectors (PODs) (Chelonia Ltd., www.chelonia.co.uk) were used to record the occurrence of harbour porpoises in the study area. As with TPODs they log the time of occurrence and duration of click, but in addition they also log the centre frequency and bandwidth. Data were processed using the CPOD.exe computer software (version 1.053), which assigns clicks to trains and then classifies these trains as being from cetaceans (distinguishing between porpoises and dolphin species), sonar or other trains sources. CPODs are the newest version of the TPOD. Unlike the TPOD, which uses an analogue processor, the CPOD uses digital waveform characteristics to select cetacean echolocation clicks, which are written to a removable Secure Digital (SD) memory card. Each CPOD was programmed to record a click limit of 4,096 clicks per minute and was set to record when the CPOD was orientated at angles of 0 to 82 degrees to the vertical. The CPODs were deployed in pairs separated by 500m (west-east) and in a water depth of approximately 14m (Figure 2)
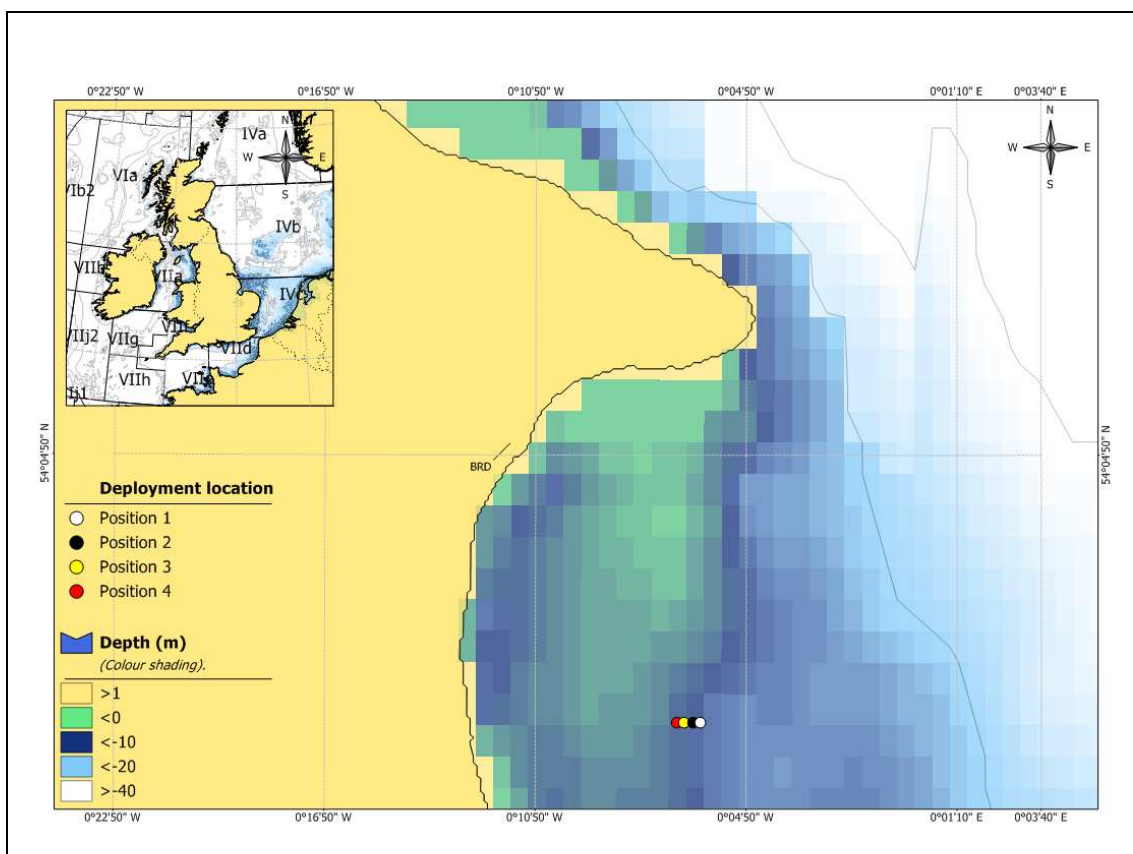
Figure 2: Locations of northern PODs deployed in Bridlington Bay

Each pair was either attached to either end of a 200m tier of nets, or was anchored at a 200m separation. Figure 3 shows a schematic of how CPODs were deployed with a net.
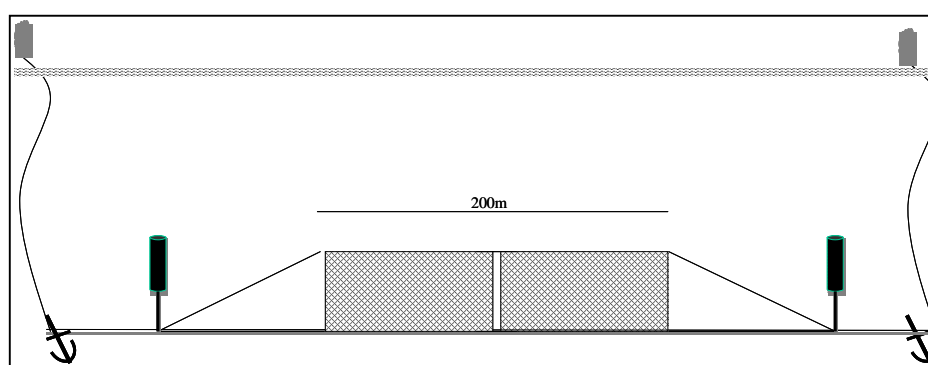
Figure 3: Schematic of POD deployment on static gillnet. PODs are not to scale.

The nets deployed in this study were the same as had been used in 2007 (See Table 1, section 2.1 for details).

Each CPOD remained in the same position for the duration of the experiment, while tiers of nets were rotated every few days between positions. Table 3 summarises net positions for the study period by deployment.

					CPOD 281(N) 264(S)	CPOD 266(N) 278(S)	CPOD 237(N) 270(S)	CPOD 268(N) 272(S)
Deployment number	Date in	Time in	Date out	Time out	Pos 1	Pos 2	Pos 3	Pos 4
1	08/07/09	09:00	11/07/09	05:00	Blank	Floats	Double	Single
2	11/07/09	08:30	15/07/09	07:30	Blank	Floats	Double	Single
3	15/07/09	11:00	19/07/09	11:30	Blank	Floats	Double	Single
4	19/07/09	14:30	22/07/09	09:00	Blank	Floats	Double	Single
5	26/07/09	09:30	30/07/09	08:00	Single	Blank	Floats	Double
6	30/07/09	11:30	03/08/09	06:00	Single	Blank	Floats	Double
7	03/08/09	09:30	07/08/09	06:30	Double	Single	Blank	Floats
8	07/08/09	10:00	11/08/09	07:30	Double	Single	Blank	Floats
9	11/08/09	10:00	15/08/09	07:30	Floats	Double	Single	Blank
10	15/08/09	10:30	20/08/09	04:30	Floats	Double	Single	Blank

Table 3: Summary of deployment times and net type. N or S in parentheses beside the CPOD identification number indicates that the CPOD was deployed at the northern or southern end of the array.

"Blanks" indicate the position where no net was deployed. In addition, data were collected on the time that nets were shot and hauled, soak duration and the fish catch per haul.

All eight CPODs were deployed for a period of 51 days. However, there was variation in the number of days logged by individual CPODs. Figure 4 shows the number of days recording for each CPOD. CPOD 268 failed to start logging at all, while CPODs 270, 264, 237 and 281 appear to have stopped logging during hauling of the nets. Data could not be downloaded until the end of the trial and therefore it was not possible to know that one of the CPODs had failed to record, and four of the CPODs had stopped recording prematurely.
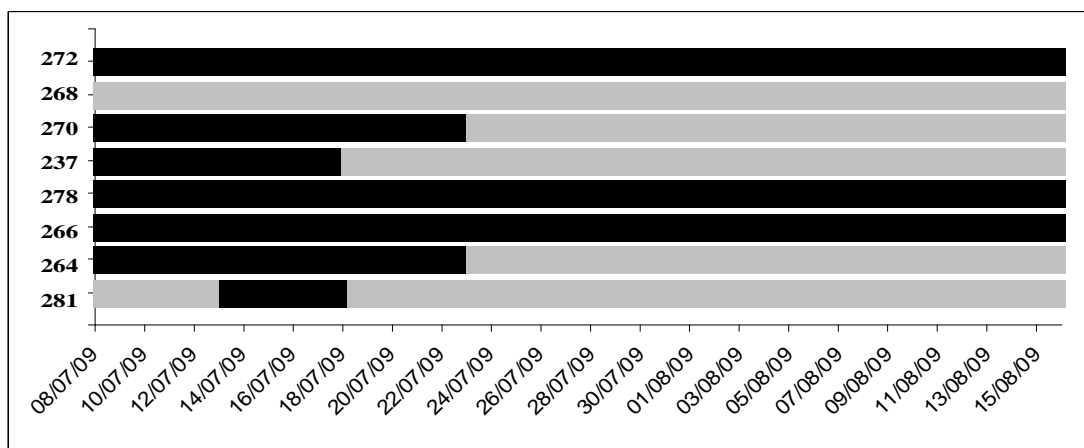
Figure 4: Black bars indicate days when each POD was recording.

5.3.1.3 CWT data – study area.

The Cornish Wildlife Trust (CWT) provided additional CPOD data which were collected during a study, conducted in 2009, investigating bycatch mitigation of harbour porpoise and bottlenose dolphins using pingers (Hardy & Tregenza 2010). Four commercial fishing vessels took part in the trial voluntarily, departing from home ports of Mevagissey, Newlyn and Helford (Fig. 5). CPODs were deployed on nets either with or without Aquamark 100 pingers. Nets were standard 10.5 inch tangle nets and each vessel worked between 8 and 10 km of nets. The exact lengths of gillnet fleets fished were not available. All nets were shot within 6 nautical miles of the shore. Soak times varied between two and five days depending on weather conditions. A single CPOD was attached to the end of a net tier. Pingered and un-pingered fleets were separated by at least 1.8 km. Data were analyzed from CPODs deployed on un-pingered tiers. Table 4 provides a summary of the data analyzed. Although skippers provided CWT with a voluntary log book in which they recorded the position and time that each tier was shot an hauled, it was clear from information on tilt angle recorded by the CPOD that there were a number of inconsistencies between the two data sets. Therefore the data on tilt angle recorded in each CPOD file was used to assign the shoot time and haul time of each tier.

Deployment	Start date	End date	No of hauls
A	08/07/09	16/07/09	1
B	30/09/09	29/10/09	4
C	01/10/09	30/10/09	7
D	09/03/10	16/05/09	17
E	05/09/09	23/09/09	5
F	02/07/09	20/07/09	5
G	28/07/09	19/08/09	5

Table 4: Summary of CWT data
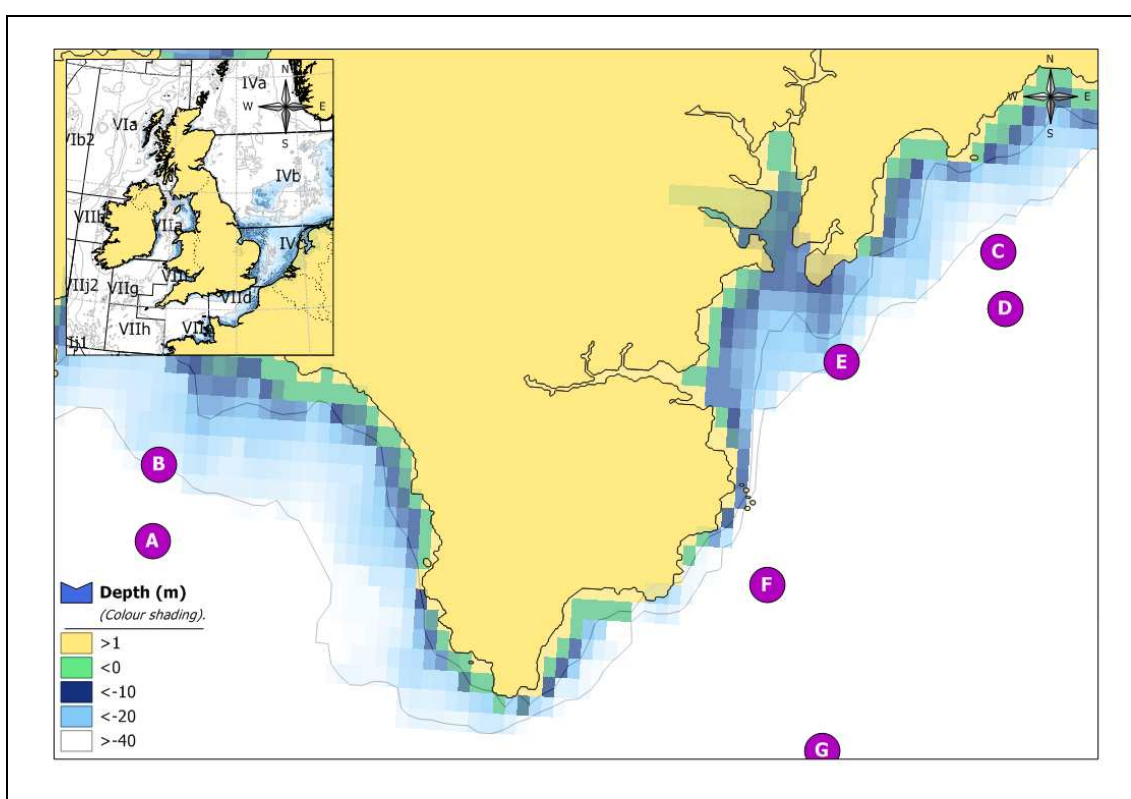
Figure 5: Locations of the first gillnet deployments for the seven CPOD files provided by CWT.

5.3.1.4 Processing CPOD data

All data recorded by CPODs was processed using version 1.053 of the CPOD.exe computer software. Visual inspection of the data showed that many harbour porpoise click trains had been misclassified by the programme as boat sonar. Figure 6 (a&b) shows screen shots taken from the CPOD.exe program, displayed by species classification, frequency logged and inter – click – intervals (ICI) of clicks in trains. The three metrics displayed by each figure are summarised in Table 5.

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Metric	Description
Species group	Different train classifications are identified by colour. Red denotes sonar, purple porpoise, orange= dolphins, green = other cetaceans,
Frequency	Logged frequencies are displayed in KHz
Inter-click-interval (ICI)	ICIs are displayed in ms.

Table 5: Description of CPOD.exe display options.

In Figure 6 a.1 data displayed by species classification shows that trains occurring within 20 seconds of each other have been classified as both arising from sonar and harbour porpoises. Investigating the frequency of the logged clicks (Fig 6 a.2) shows that both sets of trains were produced within the frequency range of clicks produced by harbour porpoise. Finally, by looking at the ICI (Fig 6 a.3) it is apparent that the intervals between clicks are variable. In contrast Figure 6b shows the same metrics displayed for logged clicks from boat sonar. The frequency of these clicks centres around 50kHz and the ICI remains constant. Therefore all trains that were classified as sonar trains were inspected by eye using these three metrics and misclassified sonar trains were then reclassified as porpoise clicks by hand.
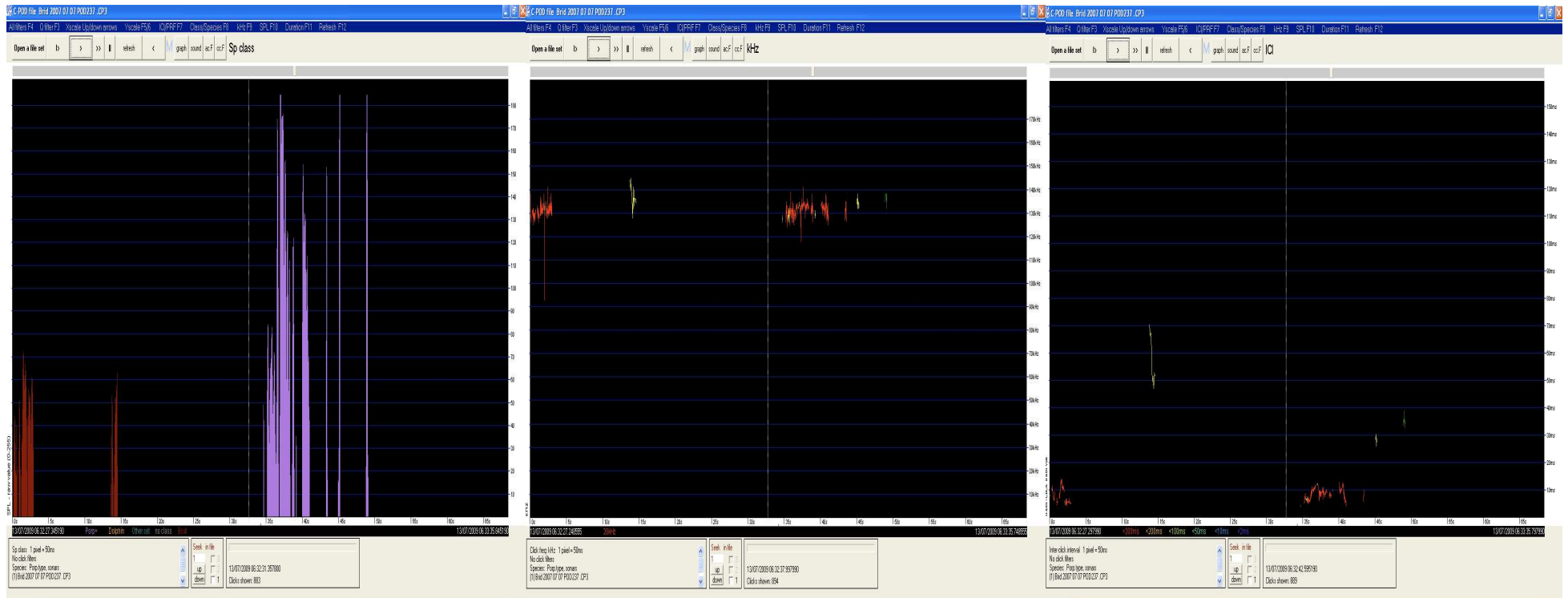
Figure 6a.1: Click trains classified by species   **Fig 6 a.2: Frequency (KHz) of logged clicks.**   **Fig 6 a.3: ICI of logged clicks.**
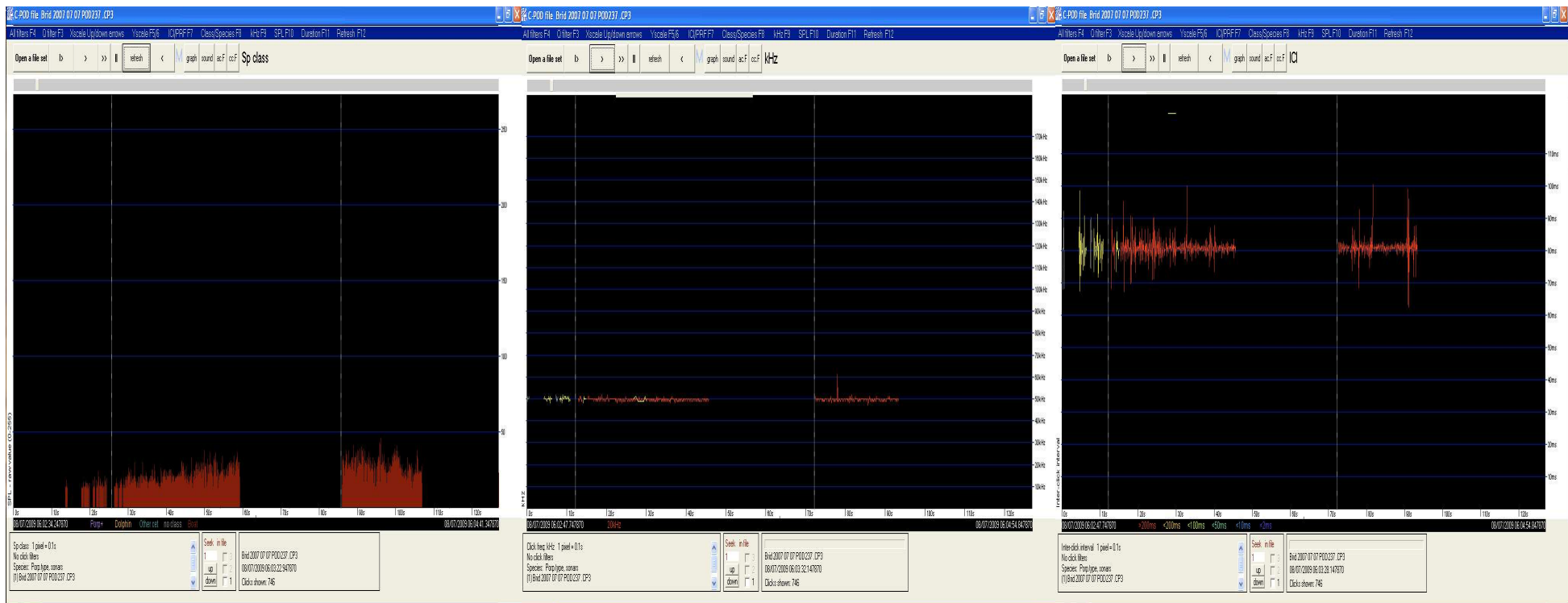
Fig 6 b.1: Click trains classified by species

Fig 6 b.2: Frequency (KHz) of logged clicks.

Fig 6 b.3: ICI of logged clicks.

The software also grades click trains according to the likelihood that they have been classified correctly as cetacean clicks into "Hi","Mod","Low" or "?" trains. These are roughly equivalent to the four train categories assigned to TPOD data ("Cet Hi", "Cet all", "Doubtful" and "??"). Previous studies using the TPODs have generally restricted analysis of cetacean click trains to those classified as "Hi" (Todd et al. 2009), or "Hi" and "Mod" (Carlstrom 2005), although some have used included "doubtful" (Carstensen et al. 2006, Philpott et al. 2007) or all four click train categories (Koschinski et al. 2006, Verfuß et al. 2007). Studies that included "??" trains in analysis assessed all trains by eye and only included those they believed had been produced by harbour porpoise. Thomsen et al. (2005) placed two TPODs in a pool with either four or two harbour porpoises and found that a high proportion of trains produced by porpoises were classified as "Lo" or "doubtful" but these trains were predominantly associated with Cet "Hi" trains. In addition, the current algorithm used to process data collected by CPODs in this study is still under development and may result in high or moderate quality porpoise click trains being classified as low quality (*N. Tregenza pers. comm.*). If click trains classified as Low probability were not produced by harbour porpoises, then including these trains in analysis would upwardly bias both the number of encounters assigned, and the length of these encounters. On the other hand, excluding Low probability trains produced by porpoises could result in encounters being separated or shortened in length, and the echolocation intensity (proportion of detection positive minutes relative to the length of the encounter) could be underestimated. Therefore a preliminary analysis was conducted to determine the impact of using Low probability click trains. Data were grouped into encounters and then the impact of using Low classification clicks on encounters was assessed.

5.3.1.5 Area of acoustic detection.

During the design of this experiment it was assumed that CPODs would have the same detection range as TPODS (~200m). However, it is possible for CPODs to detect echolocation clicks at a distance of 300m from the source (*N. Tregenza pers com*). The likelihood of logging a detection at this range will depend on the orientation of a porpoise towards the POD and the source level of the echolocation click. Figure 7 is a schematic of the theoretical acoustic detection range of CPODs in

the study. Although an overlap between PODs within a pair was expected during experimental design it was not expected that there would also be overlap between PODs separated by 500m. This overlap has implications with regards to treating click trains logged by CPODs within a treatment as independent to those logged by CPODs in neighbouring treatments.
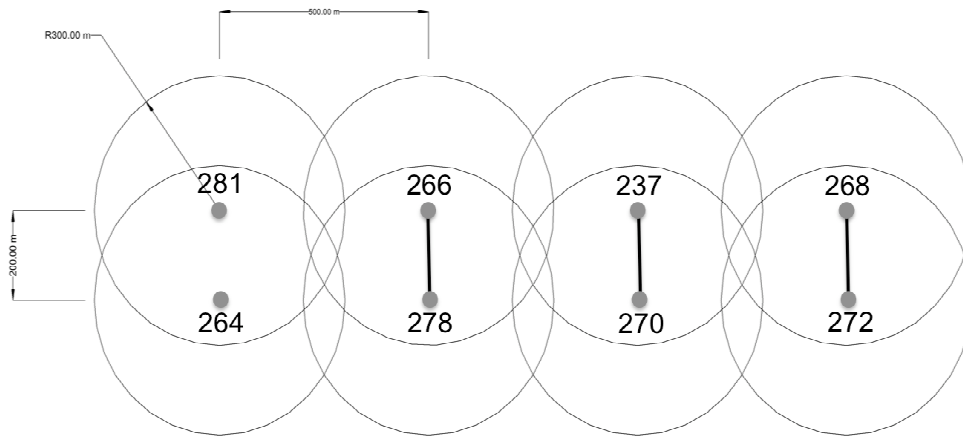
Figure 7: Schematic of the theoretical acoustic detection range of PODs in the array, denoted by the circles.

5.3.2 Echolocation metrics

There are several metrics by which TPOD and CPOD data can be analysed. The time and duration of each click train, the number of detection positive minutes (DPM), hours or days can be exported from CPOD.exe, as well as number of detections per tidal cycle. DPM can also be assigned to encounters by grouping bouts of detection positive minutes into events separated by intervening periods, of a specified length, when no clicks are detected. Figure 8 shows a schematic of the process by which clicks become assigned to encounters.
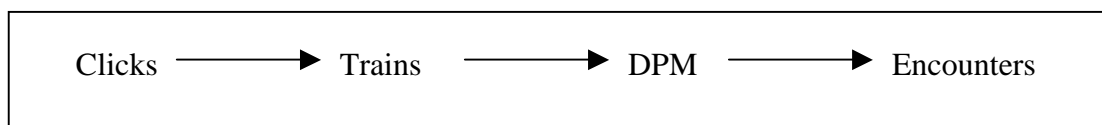
Figure 8: Schematic of process of assigning encounters logged echolocation clicks.

5.3.2.1 Detection positive minutes (DPM)

A detection positive minute is a minute in which a POD detected at least one echolocation click train. DPM were exported from CPOD.exe and used as the basic metric on which to assign encounters.

5.3.2.2 Encounters

Most TPOD studies have used echolocation rates and encounter rates to compare data collected by different TPODs, or from the same TPOD, under different treatments. Echolocation rate has been defined as the number of clicks recorded per unit of time (Cox et al. 2001, Cox & Read 2004), while an encounter has been defined as a period during which trains were detected, separated by periods of silence of 10 minutes or more. The TPOD.exe program exports the number of encounters as defined by this 10 minute silence period for specified time periods (e.g hour, day) and this has been the metric to define encounters in other studies (Carlstrom 2005, Todd et al. 2009). This function is no longer incorporated in CPOD.exe In order to compare data collected in Bridlington Bay with the results of previous TPOD studies, all DPMs were exported from CPOD.exe and then were grouped into encounters, using an algorithm written in R (R version 2.11.1), following the 10 minutes silence rule. As only 3 of the 8 CPODs recorded for the entire duration of deployment, two subsets of the data were used to assign encounters to DPMs. The first dataset was recorded by CPODs 272 and 278, which were spaced 1km apart and logged each day of the 51-day deployment. The second dataset was from 7 CPODs in the array (see Fig. 5) which all recorded during the second deployment (approx. 4 days in duration). For the 51-day data set, encounters were classified both for individual CPODs, and for the two PODs grouped together. Encounters were assigned to data collected by the seven CPODs that recorded during the second deployment, for individual CPODs, for pairs of CPODs and combining data recorded by all seven CPODs.

For the data provided by CWT, encounters were assigned to each individual CPOD file.

5.3.2.3 Encounter rate per hour

When encounter rates per hour were used in analyses, any encounters that spanned the end of one hour and the beginning of the next hour was counted only for first hour it occurred in. Encounter rates per hour were then used to estimate daily echolocation encounter rates (DEER).

5.3.2.4 Encounter intensity

While encounter length allows investigation of how long a harbour porpoise is within the detection range of a CPOD it does not give a measure of the intensity of the occurrence. Therefore the proportion of DPM within an encounter relative to the length of the encounter was used as a measure of encounter intensity.

5.3.2.5 Inter-click interval

The inter-click interval (ICI) is the time between two clicks in an echolocation train. CPOD.exe exports the details of all click trains, including the minimum inter-click interval (mICI) of that train. Currently the algorithm in CPOD.exe means that in some trains, clicks may be dropped resulting in very high minimum inter-click intervals being reported. In addition if a harbour porpoise is echolocating near to the surface the reflected clicks may be assigned to the click train and this can also affect the ICI.

5.3.2.6 Feeding buzz ratio

Todd et al (2009) identified potential feeding trains of harbour porpoises using a feeding buzz ratio, which was calculated by dividing the number of mICIs <10ms (fast trains) by those with mICIs >10ms (slow trains) for each diel phase. Although it has not been experimentally proven that fast trains recorded by PODs indicate feeding events there is evidence to show that porpoises produce buzzes, which are short series of rapid echolocation clicks, during prey capture. DeRuiter et al. (2009) deployed a modified Dtag on two captive harbour porpoise and showed that animals used echolocation clicks with slightly decreasing ICIs as they approached a prey item,

followed by fast repetition click rates of over 300 clicks per second (or echolocation buzzes) during prey capture.

5.3.3 Target range

The distance of an object, from a porpoise echolocating on that object, is a combination of the time it takes for the echo from an outgoing click to return (two-way travel time) and the length of time needed by the animal to process the information contained in the returning echo (lag time). The momentary target range of an object can therefore be calculated as:

$$D_{target} = (I - T_1)\, v/2$$

Where, D_{target}= target range (m), I= inter click interval (s), and T_1=lag time (s) and v=underwater sound velocity approx. $1500\,\text{ms}^{-1}$

Following Koschinski et al (2006), the momentary target range of the four treatments was calculated using median average and median minimum ICIs for each encounter, by substituting these values into I in equation 1.

5.3.4 Target strength of POD

The target strength of a CPOD will be affected by the strength of reflected echoes generated from the interface between the air inside the CPOD and the internal housing. Following Urick (1983) the target strength of a cylinder can be calculated as:

$$TS = 10\log\frac{aL^2}{2\lambda} \qquad \text{(Equation 2)}$$

Where a is the radius, L is the length of the cylinder and λ is the wavelength. Wavelength is calculated as:

$$\lambda = \frac{c}{f}$$

where c represents the speed of sound in water $(1,500 \text{ m s}^{-1})$ and f is the frequency. Therefore the wavelength for a harbour porpoise echolocation click with a frequency of 140kHz is $(1500/140,000) = 0.01$. The internal diameter of a CPOD is 80mm and length is 535 mm.

The estimated target strength of a CPOD is: -1.8dB. However, this does not take into account the actual volume of air remaining in a CPOD when it is deployed with a full complement of batteries.

5.3.5 Classification of tidal state

Following the methodology of Embling et al. (2010), a continuous index of tidal state was produced by assigning a score to each tidal hour. This score was calculated as a ratio between the time elapsed since the last low water to the total time between two low tides. Tidal currents were obtained using the PRODPRED programme. Table 6 summarises the classification of tidal state to tidal index.

Tidal index	Tidal state
0.01 –0.1	Low water slack
0.11 – 0.3	Flood
0.31 – 0.6	High water slack
0.61 – 0.9	Ebb
0.91 - 0	Low water slack

Table 6: Classification of tidal state.

5.3.6 Classification of diel phases.

Following the same methodology as Carlstrom (2005) and Todd et al. (2009), porpoise encounters were assigned to one of four diel phases (morning, day, evening and night). The times of sunrise, sunset and civil twilight over POD deployment periods were exported from the US Naval Observatory website (http://www.usno.navy.mil). Figure 8 shows the methodology followed as described by Todd et al (2009) to classify diel phases.

. A graphical representation of the assignment of the diel phase. The start and end of each phase was calculated using the formulae shown inside each oval, with values, and civil twilight definitions, taken from the US Naval Observatory.
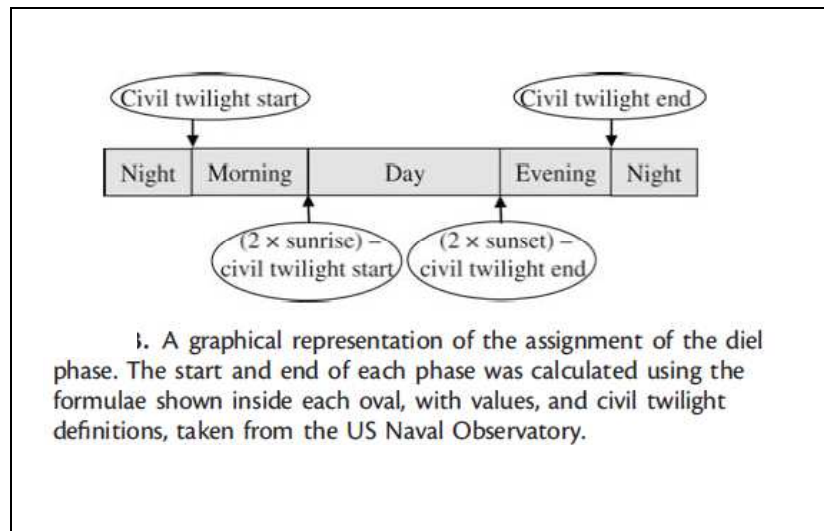
Figure 8: taken from Todd et al. (2009)

Encounters were then assigned to each diel phase using an algorithm written in R. Encounter rates per diel phase were calculated as:

$$\frac{\text{the total number of encounters in a diel phase}}{\text{mean length of the diel phase in hours}}$$

5.3.7   Statistical analysis

Data were analysed using non-parametric tests, liner models and general linear models. Goodness of fit for GLMs were assessed by investigation of residuals and Q-Q plots.

5.4 Results: Bridlington trial

5.4.1 Classification of echolocation click trains

Previous studies using TPODs have generally restricted analysis of cetacean click trains to those classified as High or Moderate. In order to fully utilize the CPOD data collected in Bridlington Bay the proportion of click trains classified as Low, as well as the probability these trains were from harbour porpoise were investigated. The number of click trains assigned to the different classification categories (High, Moderate, Low) was assessed for the two PODs which recorded over the 51 day deployment period. The majority of echolocation click trains detected by both POD 272 and 278 were classified as either High or Moderate (84% and 85% respectively).

Using all three train classifications, a total of 513 encounters were assigned to the data collected by these PODs. 34% of these encounters (n=171) were only 1 minute in length. Each of these encounters was inspected to see whether they contained High, Moderate or Low probability trains or a mix of classifications. 18% of trains recorded in these 171 encounters contained only click trains that were classified as Low probability. Each of these Low probability encounters (n=31) was visually inspected to assess whether Low categorized trains making these encounters were likely to have come from porpoises. Only 1 detection positive minute (DPM) contained boat sonar, which had been misclassified as a porpoise click train. Of the remaining 30 DPM 77% were visually determined to have most probably have come from a porpoise whilst 23% were doubtful but still likely to have come from a porpoise.

As previously stated, the inclusion of Low probability click trains, which were not produced by a porpoise, could upwardly bias the estimate of encounter lengths. Therefore, the number of encounters where a period of two minutes or longer period had passed between the detection of a Low DPM and a preceding or subsequent High or Moderate DPM was assessed. 42 encounters contained Low probability DPM which did not occur within the minute directly proceeding or subsequent to a High or Moderate probability DPM. A subset of 20 of these encounters was assessed by eye and after visual inspection all Low probability DPM in these encounters were

considered to be from harbour porpoise clicks. Therefore there was no reason to exclude Low probability trains from further analysis and all three train classification levels (High, Moderate and Low) were used to assign click trains to encounters.

5.4.2 Investigation of harbour porpoise occurrence and echolocation behaviour in the study area.

5.4.2.1 Daily echolocation encounter rate (DEER)

Daily echolocation encounter rates were calculated for PODs 272 and 278. Harbour porpoise echolocation click trains were logged on all days during the deployment period, except for the 16th of August when no harbour porpoise click trains were recorded by POD 278.
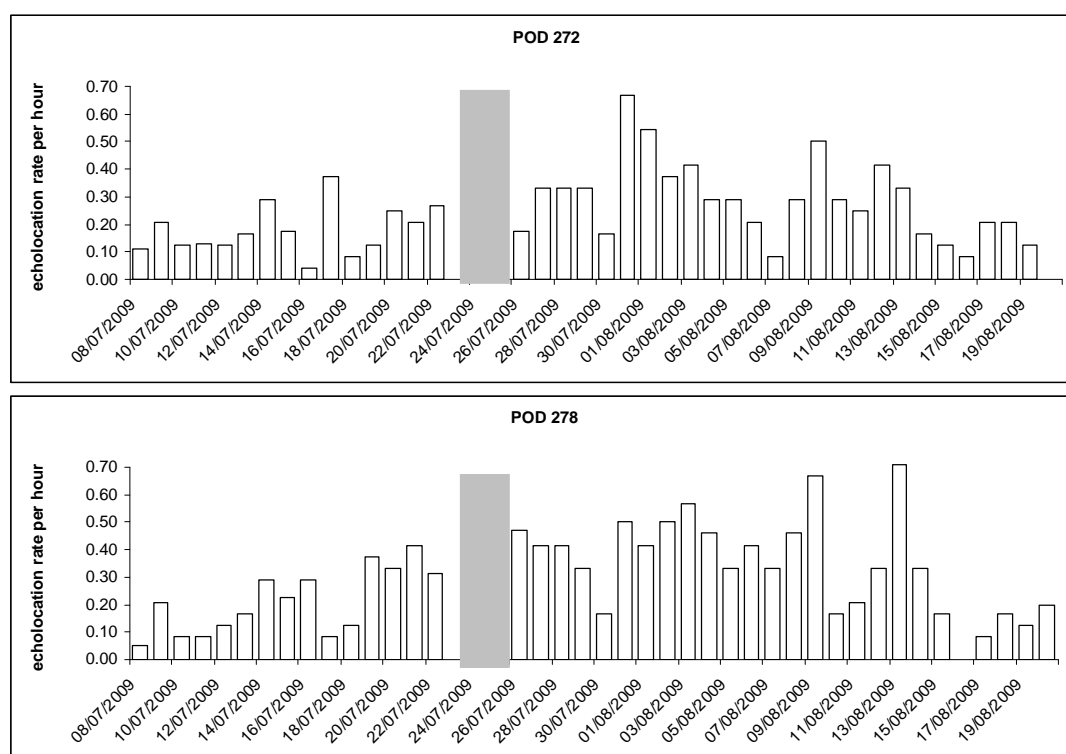
Figure 9: Daily mean echolocation encounter rate per hour for POD 272 and 278. Grey shading indicates when nets were removed from the water during spring tides.

Figure 9 shows the daily mean echolocation encounter rates (DEER) for PODs 272 and 278 over the duration of the study period. DEER could be approximated to a normal distribution (Lilliefors test for normality: POD 272, n=41, p=0.09; POD 278, n=41, p=0.22). Linear regression analysis indicates that there was no trend in daily

echolocation encounter rates throughout the study period for either POD 272 (r^2 =-0.015, p=0.53) or POD 278 (r^2 =0.006, p=0.27). Table 7 summaries the number of encounters and mean encounter rates per hour detected by each POD.

POD	Total number of encounters	Mean enc rate per hour	S.E.
272	233	0.25	0.03
278	280	0.30	0.03

Table 7: Summary of encounters and encounter rate by POD

5.4.2.2 Comparison of encounter rates during the study period.

Overall, POD 278 had significantly more encounters per hour than POD 272 during the study period (GLM, poisson on encounter rate per hour, p=0.04). Figure 10 shows the encounter rate per hour for each deployment for both PODs and it is clear that encounter rates per hour were not always highest in POD 278. Neither POD recorded more than 3 encounters within a single hour, and no encounters were detected in the majority of hours that the PODs were recording (POD 272:79%, POD 278: 75%).
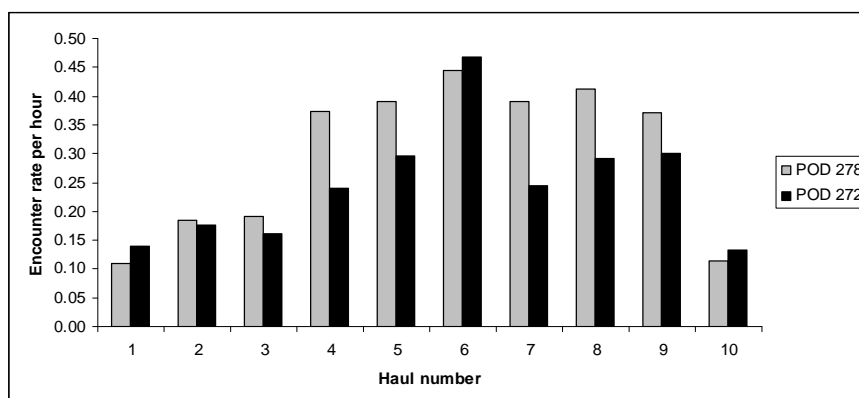
Figure 10: Encounter rate per hour for each POD and each deployment.

5.4.2.3 Encounter length.

Encounter lengths recorded by both PODs, over all deployments, ranged from a minimum of 1 minute (the minimum encounter length possible when encounters are assigned to DPM) to a maximum of 44 minutes. Average encounter lengths for PODs 272 and 278 were 4.9 minutes (S.D. 5.7) and 5.1 minutes (S.D. 5.4), respectively. 33% of all encounters recorded by both PODs were 1 minute in length (n=171). 46%

of these were recorded on POD 272 while 54% were recorded on POD 278. Figure 11 shows a box plot of encounter lengths for each POD and each deployment.
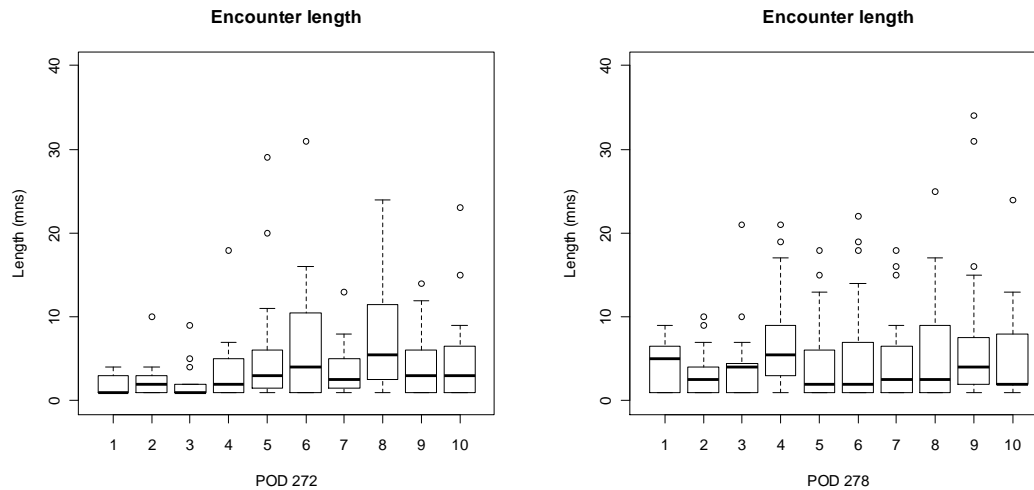
Figure 11: Box-plots of encounter lengths recorded by POD 272 & 278 during each deployment.

5.4.2.4 Encounter rate with tidal phase

Figure 12 shows the mean echolocation encounter rate per tidal phase. The distribution of echolocation encounter rate with tide index was non-normal (Lilliefors test for normality, $p < 0.001$) but homogeneous (Bartlett test, $p > 0.5$). There was no significant difference in echolocation encounter rate by tidal index (Kruskal-Wallis, one-way ANOVA on ranks, chi-squared = 8.5285, df = 9, $p > 0.1$).
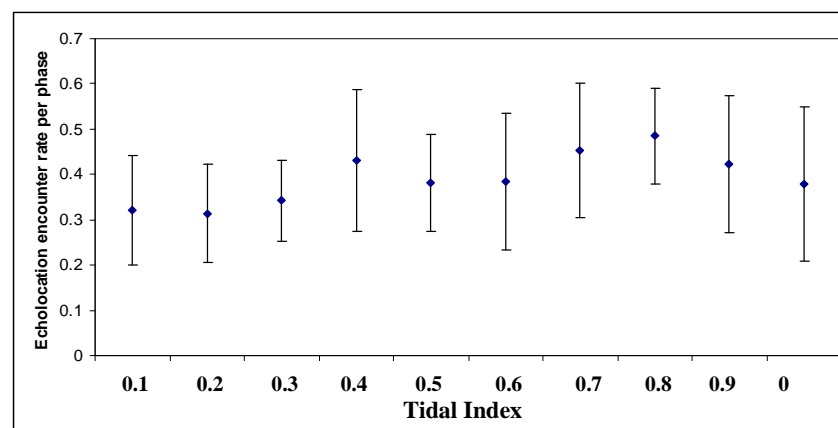
Figure 12: mean echolocation encounter rate per tidal phase with standard errors.

5.4.2.5 Encounter rate with diel phase

Encounter rates for each diel phase were calculated individually for POD 272 and POD 278. However, plots of the mean and variance in encounter rates recorded by each POD during each diel phase showed that the PODs could be grouped together. Mean encounter rates were highest during the day and morning (Table 8)

Phase	Mean phase length (hrs)	S.E	Mean encounter rate per diel phase	S.E
morn	1.5	0.0	0.53	0.12
day	14.3	0.1	0.62	0.06
eve	1.5	0.0	0.36	0.10
night	6.7	0.1	0.37	0.04

Table 8: Mean phase length and encounter rate by diel phase

Echolocation encounter rates per diel phase were not normally distributed (Lillifors test for normality, n=159 , p <0.001) or homogenous (Bartlett's K-squared = 51.2094, df = 3, p <0.001). Therefore a recently published methodology for comparing multiple means under non-normality, unbalanced group sizes and heteroscedasticity was used to test if there was any difference in echolocation encounter rate with diel phase (Herberich et al. 2010). Results of this general linear hypothesis testing using Tukey's all pairwise comparisons showed that there were significantly more encounters per hour during the day than the night (p=0.00282).

5.4.2.6 Echolocation click trains and potential feeding trains

A total of 4,028 individual echolocation click trains was recorded by POD 272 and 278 combined. Of these 10% fell into the defined <10ms minimum inter click interval (mICI). Figure 13 shows the distribution of mICI for each haul by POD. The red line indicates mICI less than 10ms in length, which Todd et al. (2009), using TPODs, attributed to feeding buzzes. The minimum ICI recorded by either POD was 1.05ms and the maximum was 282.35ms.
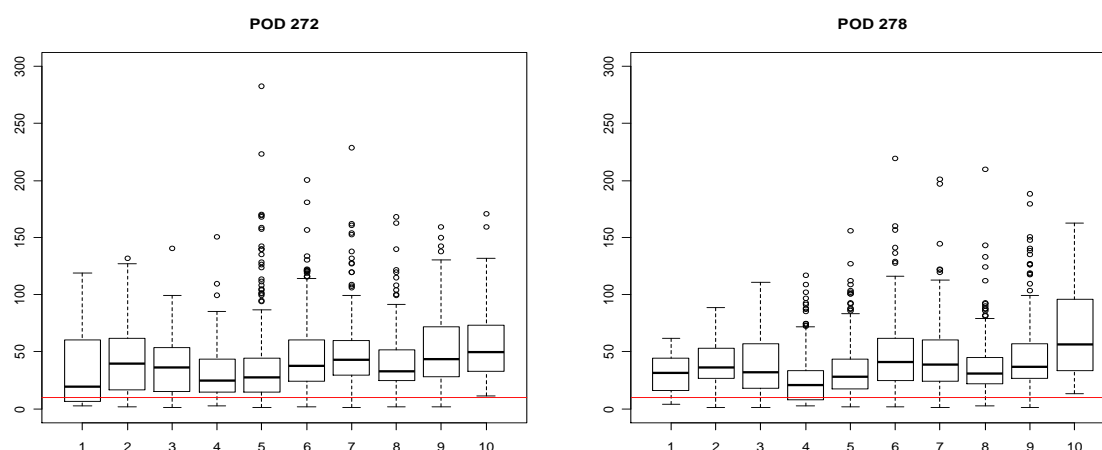
Figure 13: Boxplots of minimum ICI by haul for each POD.

Minimum ICIs were then grouped according to diel phase. Table 9 provides a summary of mean ICIs and the percent of ICIs in each phase which were less than 10 ms, while Figure 14 shows the distribution of minimum ICIs per diel phase. Minimum ICIs were not normally distributed (Lilliefors D = 0.1141, p< 0.001) and heterocedastic (Bartlett Bartlett's K-squared = 24.8847, df = 3, p < 0.001).

Phase	N	Mean ICI (ms)	% trains mICI <10ms
Morning	231	49	5.2%
Day	2895	40	10.4%
Evening	236	46	2.5%
Night	666	43	15.2%

Table 9: Summary of ICI by diel phase.
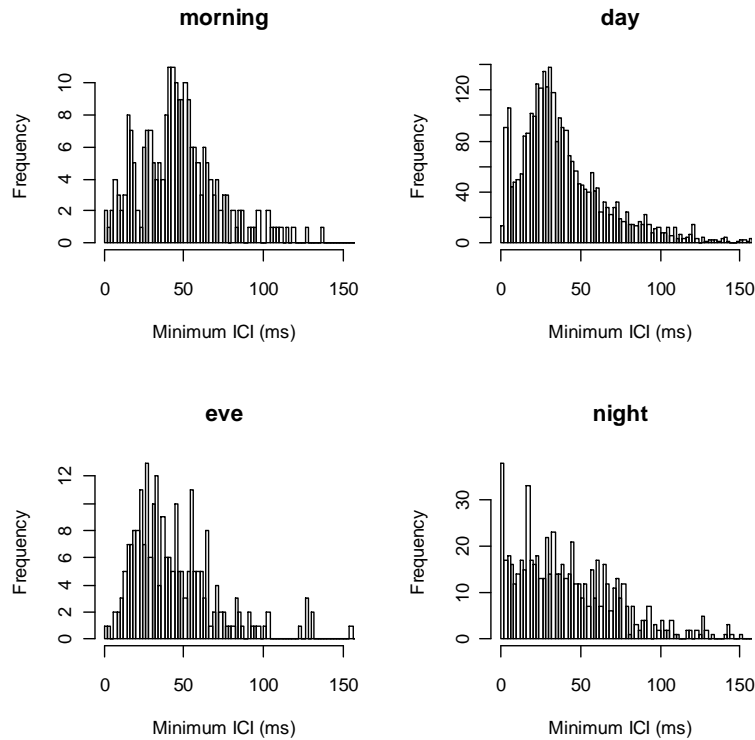
Figure 14:Distribution of mICI by diel phase.

Results of general linear hypothesis testing using Tukey's all pairwise comparisons showed that mICIs were significantly lower in the day compared to the evening ($p<0.05$) or morning ($p<0.001$), and significantly lower in the night compared to the morning ($p<0.05$).

5.4.3 Differences in harbour porpoise presence and echolocation behaviour during different treatments.

The experiment in Bridlington was designed to allow contemporaneous comparisons of harbour porpoise echolocation behaviour for each of the four treatments (no net, Single net, Double net, Float net). As a number of CPODs failed to record such a comparison was not possible and analysis was restricted to comparing pairs of treatments where PODs recorded simultaneously. These treatment pairs were No net – Double net (Deployment 5,6,9 &10) and Single net – Float net (Deployment 1-4, 7 & 8). As treatments were rotated between the four deployment locations in Bridlington

Bay, each of the two PODs that recorded for the entire duration of the 51 day experiment were deployed with each treatment for a minimum of four deployments.

5.4.3.1 Comparison of encounter rates in the presence of absence of a fishing net and between nets with different float lines

In order to assess whether harbour porpoises are attracted to gillnets, data recorded by PODs 272 and 278 during deployments 5,6, 9 and 10 were used to assess if there was any difference in the number of encounters recorded per hour when the Double net was or was not present. Table 10 shows the treatment for each POD during these deployments.

POD	Deployment no.	Treatment
272	5 & 6	Double net
	9 & 10	No net
278	5 & 6	Double net
	9 & 10	No net

Table 10: Summary of treatments with each POD by deployment number.

The number of encounters per hour was modelled using a GLM with Poisson error distribution and log link function. Results showed no significant difference between treatments in any of the four deployments (Deployment 5: $p > 0.1$,Deployment 6: $p > 0.1$, Deployment 9: $p > 0.1$ and Deployment 10: $p > 0.1$).

The same model was run, this time comparing encounter rates per hour between the net with the single float line and the net with polypropylene floats (see Table 11 for deployment details). During deployments 1-4 encounter rates were lower in the vicinity of the single float line net but not significantly ($p > 0.1$). In contrast, during hauls 7 and 8 encounter rates were significantly lower in the vicinity of the net with polypropylene floats ($p < 0.05$).

POD	Deployment no.	Treatment
272	1 - 4	Single float line
	7 & 8	Polypropylene float line with floats
278	1 - 4	Polypropylene float line with floats
	7 & 8	Single float line

Table 11: Summary of treatments with each POD by deployment number.

5.4.3.2 Investigating differences in encounter length and intensity in the presence or absence of gillnets and nets with different float lines

Although there was no significant difference in the encounter rates recorded by PODs deployed with the Double net or with and no net, encounter rate only captures occurrence of harbour porpoise detections and not the length of encounters around different treatments. Therefore data on encounter lengths for hauls where both the Double net and no net were deployed were compared. As the variance in encounter length was large, a GLM model with quasi-Poisson distribution (with variance equal to mean squared) was constructed. There was no significant difference in encounter lengths with a net (POD 278), or without a net (POD 272), during deployments 5 and 6 (p= 0.388 and p=0.226). Likewise there was no significant difference in encounter lenths with a net (POD 272) or without a net (POD 278) during deployments 9 and 10 (p= 0.07 and p=0.775).

While encounter lengths provide a measure of the temporal occurrence of harbour porpoises they do not provide a measure of the intensity of the occurrence (i.e. the proportion of DPM relative to the length of encounter). By definition an encounter must always start with a detection positive minute, and will always end with a last detection positive minute directly prior to a period of 10 minutes or more during which no further detections are logged. Therefore, the first and last DPM logged in encounters were removed. As a result encounters of 1 and 2 minutes in length were excluded from the analysis. A binomial GLM model with logit link function was constructed to test the proportion of DPM within an encounter with a net or not net. There was no significant difference (p=0.9) between treatments. Figures 15 and 16

show encounter lengths and encounter intensity collected by PODs deployed with or without a net.
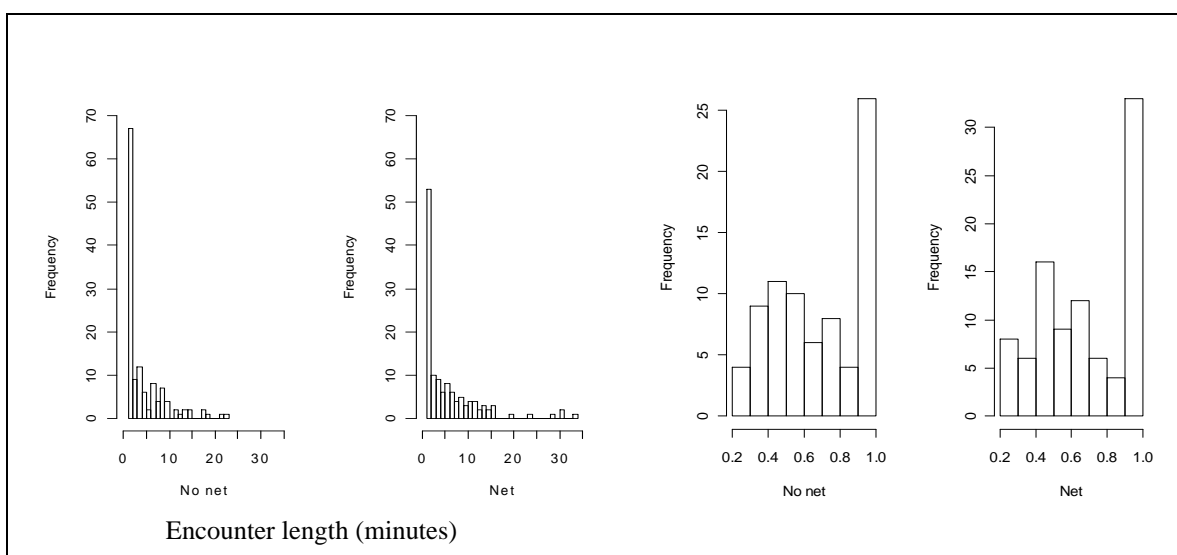
Figure 15: Histogram of encounter lengths by treatment type.

Figure 16: Histogram of the proportion of DPM relative to encounter length.

The same methodology was used to compare encounter length and intensity for deployments with the single float line net and the net with polypropylene floats. Encounters were significantly longer in the presence of the net with polypropylene floats during deployments 1-4 ($p<0.05$) but there was no significant difference in the encounter lengths recorded at the two nets during deployments 7 and 8 ($p>0.05$). Likewise, there was no significant difference in encounter intensity between the two treatments ($p>0.1$). Figures 17 and 18 show encounter lengths and encounter intensity collected by PODs deployed with the Single float line net and the net with additional floats.
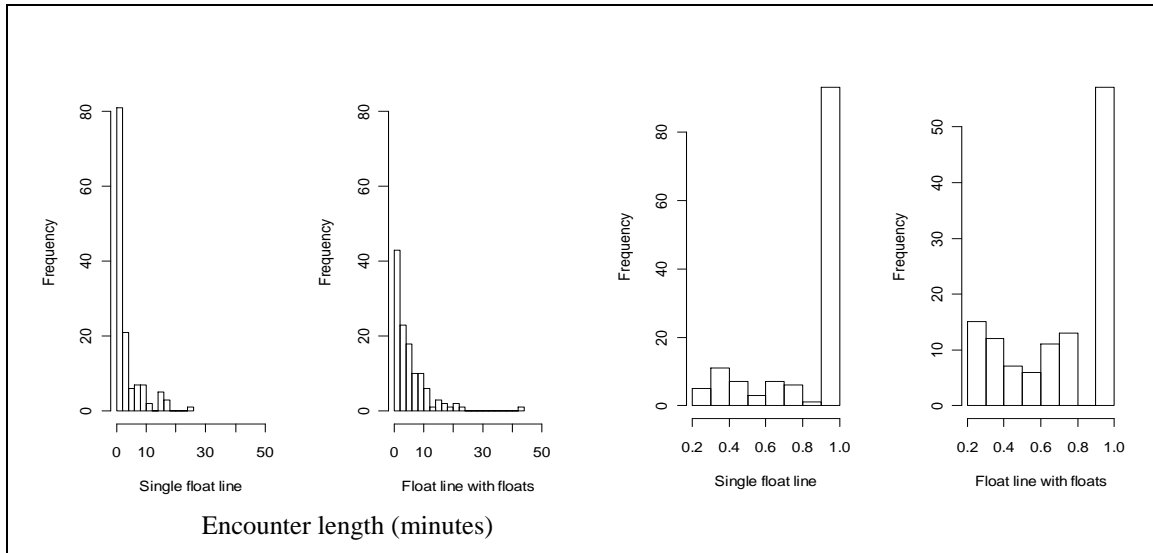
Figure 17: Histogram of encounter lengths by treatment type.

Figure 18: Histogram of the proportion of DPM relative to encounter length.

5.4.3.3 Investigating mICI in the presence and absence of gillnets and with different types of nets.

Koschinski et al (2006) analyzed the distribution of click intervals recorded on a TPOD to infer the target range of an experimental gillnet to free ranging echolocating harbour porpoises. Figures 19 and 20 show the distribution of average ICIs per click train for deployments (5,6,9 & 10) where the two treatments were a net and no net and where the treatments were a net with a single float line versus a net with polypropylene floats. In order to obtain 95% confidence intervals (Fig 20) encounters were re-sampled with replacement, as click trains within an encounter are likely to be correlated.
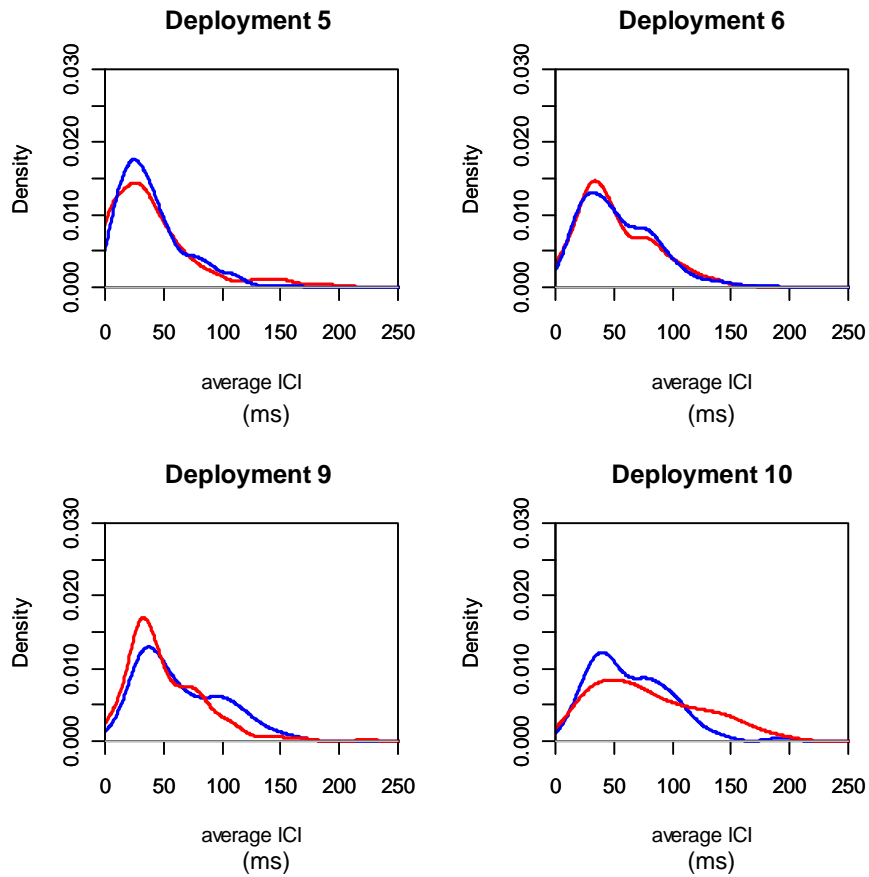
Figure 19: distribution of average ICIs in ms for PODs deployed in the presence and absence of a gillnet. Red line = net present, blue line = net absent.
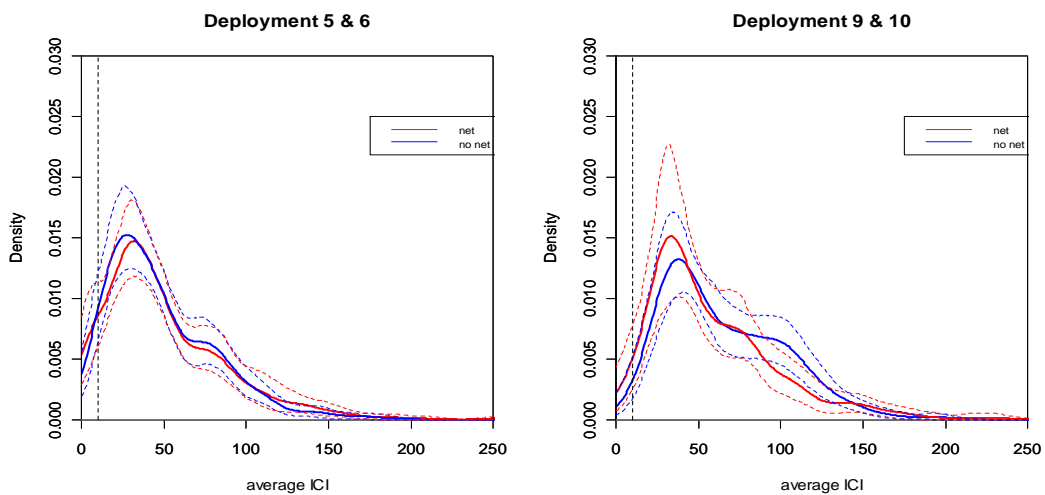
Figure 20 : combined distribution of minimum ICIs in ms for POD deployments in the presence and absence of a gillnet. Dashed lines represent 95% confidence intervals. Red line = net present, blue line = net absent.

The same methodology was used to produce density plots of average ICIs (ms) recorded in the presence of a net with a single float line, and a net with polypropylene floats for all 6 deployments of these treatments (Fig. 21) and for deployments 1-4 combined and 7 and 8 combined (Fig. 22).

There was no significant difference in the pooled distributions of either average ICIs (Kolmogorov-Smirnov test; D= 0.0515 p>0.1) or minimum ICIs (Kolmogorov-Smirnov test; D= 0.0526 p>0.1) for deployments when a net was or was not present. Likewise there was no difference in the pooled distributions of average ICIs (Kolmogorov-Smirnov test; D= 0.0514 p>0.1) or minimum ICIs (Kolmogorov-Smirnov test; D= 0.0541 p>0.1) between a net with floats or with a single float line.
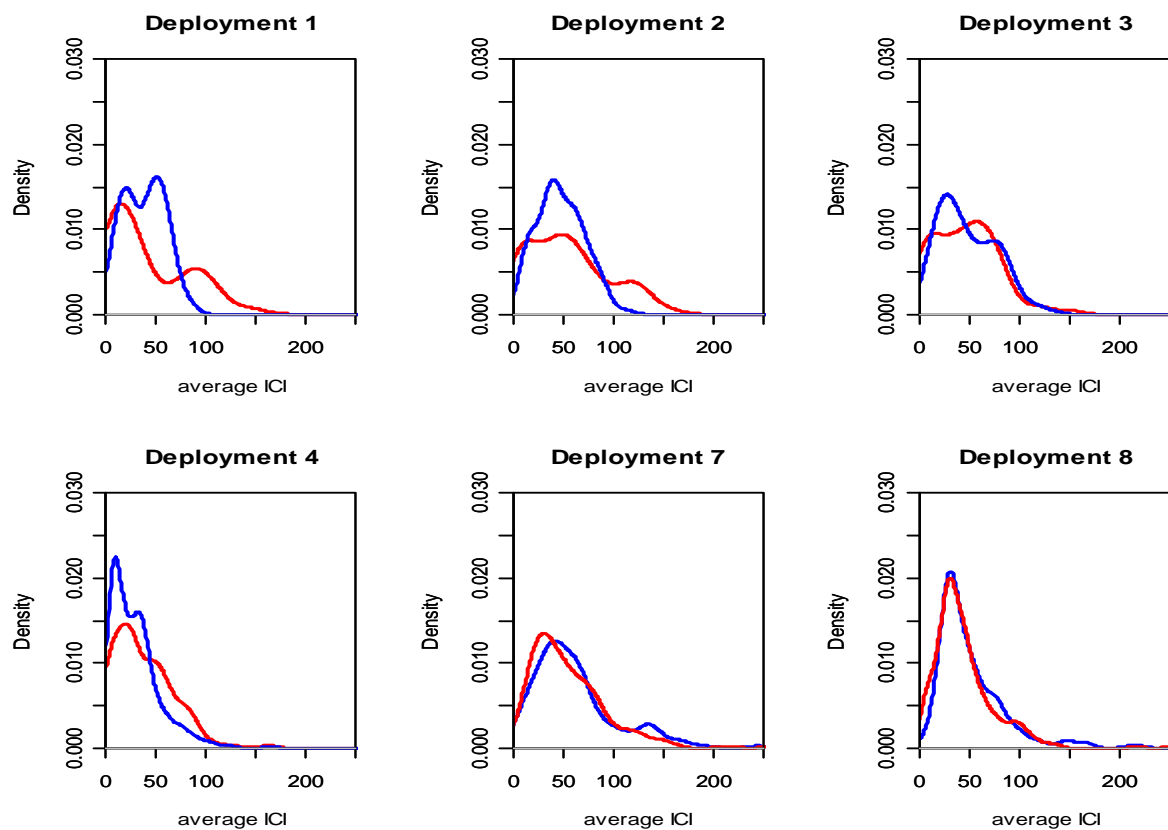
Figure 21: distribution of average ICIs in ms for PODs deployed in the presence of two gillnets with different float lines. Red line = single float line, blue line = polypropylene float line.
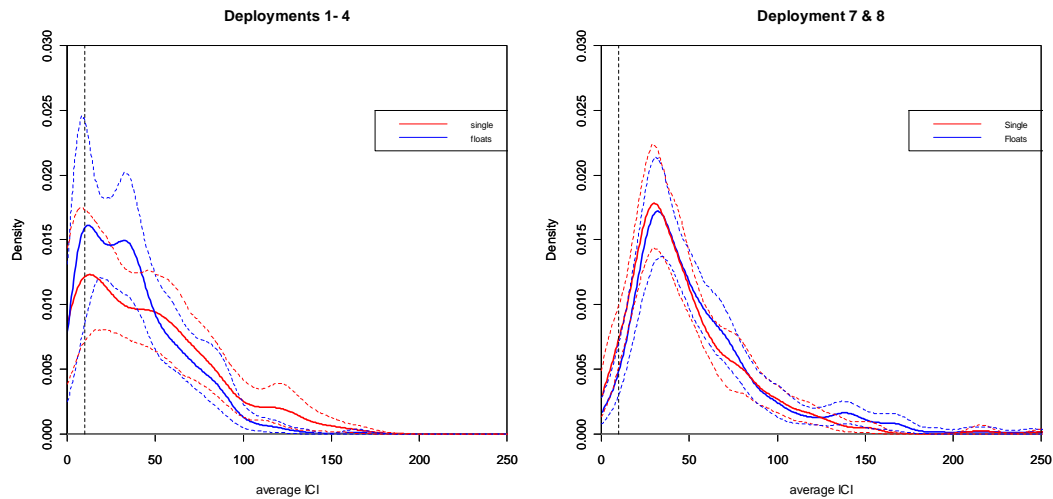
Figure 22: combined distribution of average ICIs for PODs with nets of two float line types. Dashed lines represent 95% confidence intervals. Red line = single float line, blue line = polypropylene float line.

Table 12 summaries the median average ICI and median minimum ICI per click train by treatment type.

Deployment	POD	Treatment	Median average ICI (ms)	Median minimum ICI (ms)
5&6	278	No net	39	34
	272	Net	40	33
9 & 10	278	Net	46	40
	272	No net	55	45
1-4	278	Polypropylene float line with floats	33	38
1-4	272	Single float line	40	31
7 & 8	278	Single float line	40	33
7 & 8	272	Polypropylene float line with floats	44	37

Table 12: Summary of median average ICI and median minimum ICI for each treatment.

If median average and median minimum ICIs do represent the target range at which harbour porpoises detected the four different treatments, the lack of significant difference in these metrics, when a net was or was not present, suggests that harbour

porpoises in this study area may have been targeting their echolocation clicks at the CPODs rather than at the nets.

5.4.3.4 Proportion of fast to slow click trains in encounters in the presence and absence of a net, and for nets with different float lines.

Only 3% of encounters had a feeding buzz ratio (FBR) >1 as defined by Todd et al (2009). Figure 23 shows the proportion of encounters with FBR>1 by net type. The highest number of encounters with an FBR>1 were recorded at the net with the single float line. However, this amounted in total to only 6 encounters.
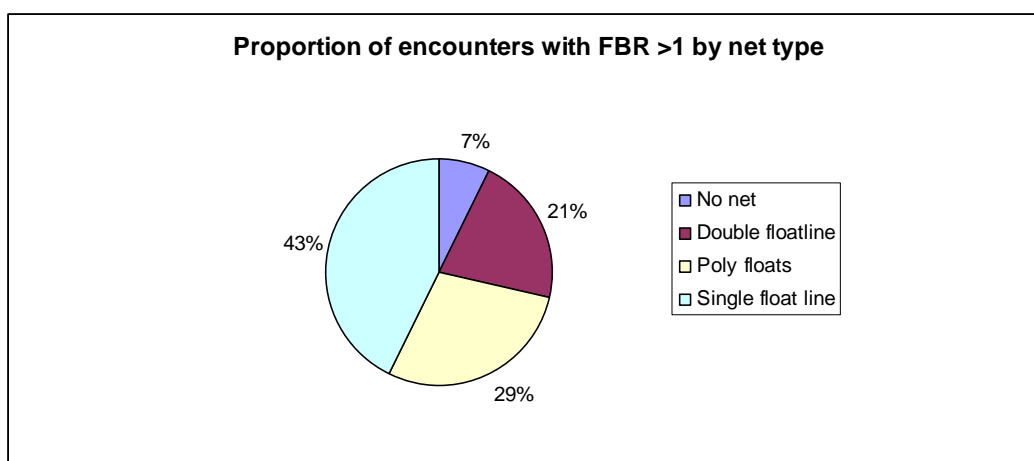
Figure 23: Proportion of encounters with FBR>1 by treatment type.

For their analysis Todd et al. (2009) grouped all click trains recorded in a diel phase to calculate FBRs. In order to investigate the relationship between the proportion of fast click trains and encounter length and to investigate whether harbour porpoises might be foraging in the presence of nets, or inspecting nets more closely, the proportion of fast (mICI <10ms) to slow (mICI ≥ 10ms) echolocation click trains within each encounter was calculated.

These proportions were modelled using a binomial GLM with logit link function. Encounters with a greater proportion of fast echolocation click trains were assigned the value 1 while those with a greater proportion of slow click trains were assigned the value 0. Results showed that there was a significant positive relationship between

the proportion of fast trains in an encounter and encounter length (P=0.0132). When data were restricted to deployments where one of either POD 272 or 278 was not deployed with a net, there were significantly more fast to slow trains per encounter, in the presence of a net than when there was no net present (p<0.01). However, there was no significant difference in the proportion of fast to slow trains recorded by PODs deployed with the net with a single float line or the net with additional floats (p>0.5).

1.5% of all encounters recorded when no net was present had a higher proportion of fast to slow trains. In contrast when a net was present the percent of encounters with a higher proportion of fast trains ranged from 4.7% (double float line net) to 12.6% (single float line net). If fast click trains represent foraging behaviour, then this result suggests that porpoises stayed longer in the array when engaged in foraging.

5.4.4 Harbour porpoise movement around static gillnets.

5.4.4.1 Encounters during deployment number 2

Deployment 2 was the only period when 7 of the 8 PODs were recording, and therefore provided the only data where it would be possible to compare echolocation metrics between the four treatments concurrently. Table 13 provides a summary of encounters on each POD during deployment 2 which was the only period when 7 of the 8 PODs were recording. The overall encounter rate and average encounter rate per hour between individual PODs varied, but this variation was not consistent within individual treatments. This may indicate that animals generally did not move along nets in the array.

Treatment	North/South	POD	No. of encounters	Average encounter rate per hour
No net	North	281	13	0.26
No net	South	264	17	0.17
Floats	North	266	15	0.15
Floats	South	278	18	0.18
Double	North	237	35	0.36
Double	South	270	11	0.11
Single	North	268	NA	NA
Single	South	272	17	0.18

Table 13: Summary of encounters by PODs and positions

Table 14 summarises the mean encounter length in minutes recorded by each POD and mean and quartile encounter lengths are plotted in Figure 24.

Treatment	Pod	Mean encounter length (mins)	S.E.
No net	281	7.6	1.4
No net	264	3.5	0.5
Floats	266	4.1	0.9
Floats	278	3.5	0.7
Double	237	8.3	2.0
Double	270	2.5	0.6
Single	272	2.5	0.5

Table 14: Mean encounter length recorded by each POD.
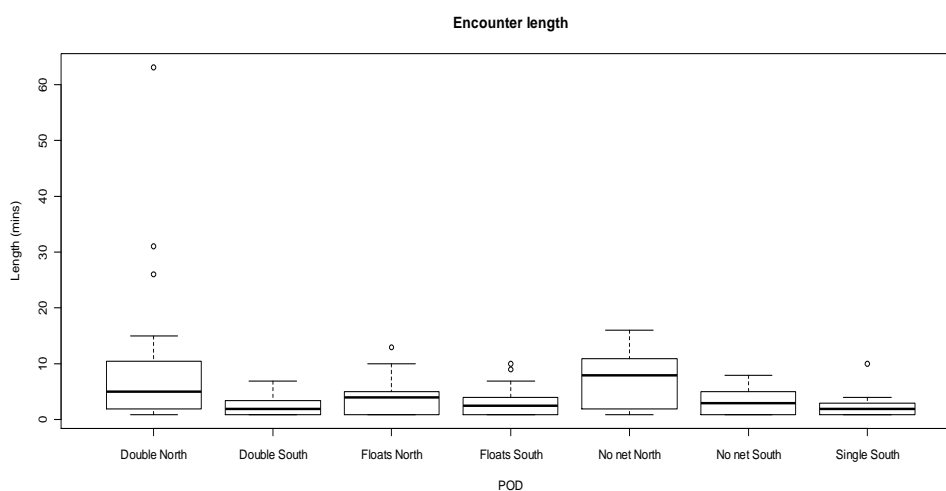
Figure 24: Box-plot of encounter length by individual POD, Haul 2.

In order to determine the amount of overlap in detections between PODs in the array, DPMs were analysed to see how many minutes were shared between pairs of PODs in the same treatment and with PODs in other treatments. Of the 366 DPM recorded by PODs in the array over the 4 day deployment, 9% were recorded on 2 PODs (n=34) and 3% were recorded simultaneously on three PODs (n=11). As PODs within treatments are separated by only 200m we would expect that DPM recorded simultaneously would be most likely within pairs, however, only 50% of these shared DPM occurred between PODs in the same treatment. Of the 11 DPM recorded simultaneously on 3 PODs, 9 occurred between 2 PODs within a treatment and a POD

in a treatment 500m away. The remaining 2 simultaneously recorded DPM occurred in a single POD from three different treatments (Fig. 25). It is likely that those DPM, which were recorded on 3 PODs simultaneously, occurred when more than 1 harbour porpoise was present in the array.
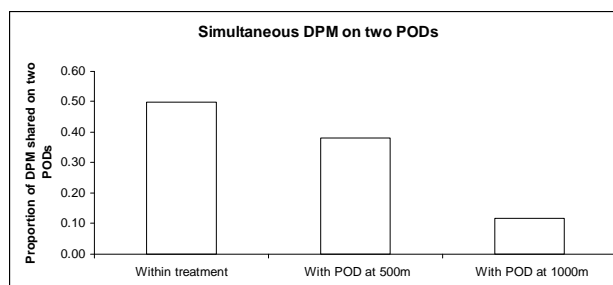
Figure 25: Proportion of DPM recorded by two PODs.

As 50% of simultaneously recorded DPM were recorded by pairs of PODs within a treatment, data were grouped for each pair of PODs, and new encounters were assigned to these data. For example if a DPM was recorded by the POD deployed on the northern end of the Double net, and a DPM was recorded in the subsequent nine minutes by the POD deployed on the southern end of the net, these DPM would be grouped in the same encounter. Table 15 shows the encounter rate per treatment for pairs of PODs.

Treatment	No of encounters	Encounter rate per hour
No net	30	0.20
Floats	19	0.10
Double	44	0.23
Single	17	0.18

Table 15: Summary of encounters grouped by pairs of PODs, Haul 2.

21% of all encounters (n=23), assigned to pairs of PODs within a treatment, were found to share a simultaneous DPM with at least 1 encounter that had occurred in another treatment. Of the 25 separate DPM that were recorded simultaneously by PODs in different treatments, 72% were recorded by PODs in neighbouring treatments, 20% were recorded by PODs which were separated by at least one other treatment and 8% were recorded simultaneously by PODs in three different treatments. As the same click train could be picked up by PODs in neighbouring

treatments, this means there is psuedo-replication of data recorded between treatments, and it is not possible to test whether there is a significant difference in encounter rates between these treatments.

Figure 26 shows a boxplot of encounter lengths by treatment type. Mean encounter rates were similar between treatments. Shortest encounters were in the single treatment that only contained 1 POD during this deployment.
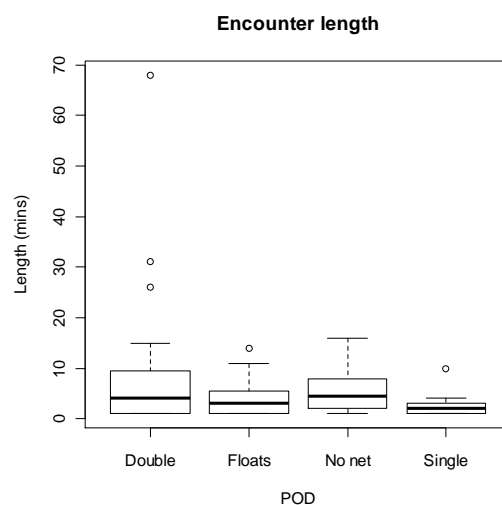
Figure 26: Box-plot of encounter length for pairs of PODs, Haul 2.

81% of these encounters lasted less than 10 minutes in length and 29% were only 1 minute in length. Encounters from each pair of PODs were then investigated to see whether encounters always contained both PODs in a pair, or only contained DPM on one of the two PODs. Figure 27 shows the proportion of encounters recorded on either the North, South or on both PODs in each pair for deployment 2. The single headline net only had one POD recording.
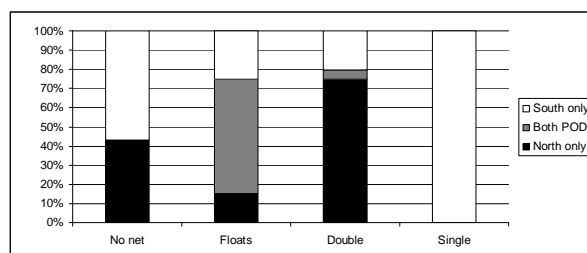
Figure 27: Proportion of encounters in Haul 2 recorded on one or both PODs in a pair.

This graph would suggest that, when there is no net, animals rarely move from one POD to another within an encounter. However, data from deployment 2 are limited and the northern POD without a net did not start to record for the first 40 hours of deployment. Therefore the same methodology was applied to a pair of PODs (POD 266 & POD 278), that recorded for the duration of the 51-day deployment (Fig. 28).
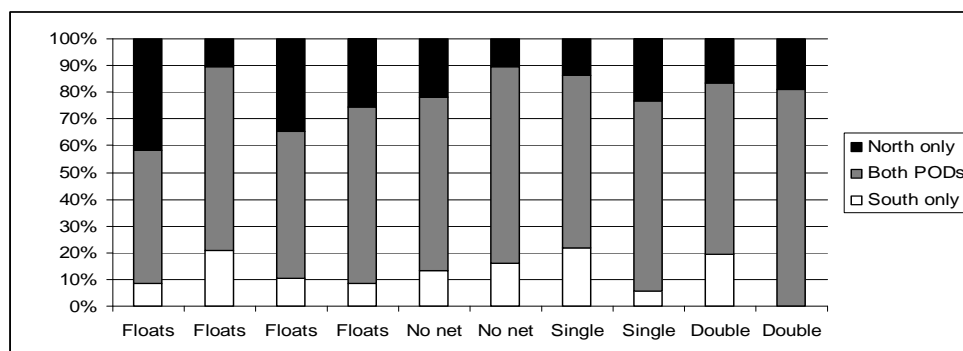
Figure 28: Proportion of encounters by treatment for pair of PODs recording for 51 days.

These data suggest that between 50-80% of all encounters are recorded by a pair of PODs regardless of the treatment.

5.4.4.2   Movement around the array.

Due to the likelihood of PODs in different treatments detecting the same porpoise, the data were then reanalysed and encounters were assigned to data pooled from all PODs in the array This resulted in a total of 75 encounters during deployment 2. Figure 29 shows the frequency of encounter lengths from this pooled data set and indicates that the majority of encounters contain only 1 DPM, all of which were recorded on a single POD. Figure 30 shows a box plot of encounter length and the number of different PODs that logged click trains within that encounter. There is a general trend that as the length of an encounter increases so does the number of individual PODs that recorded DPM within that encounter. This suggests that the longer a harbour porpoise stays in the vicinity of the array, the more it moves around the array.
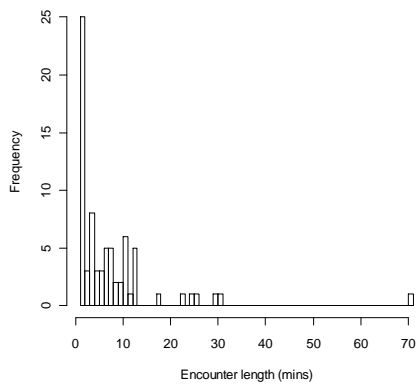
Figure 29: Histogram of encounter lengths.
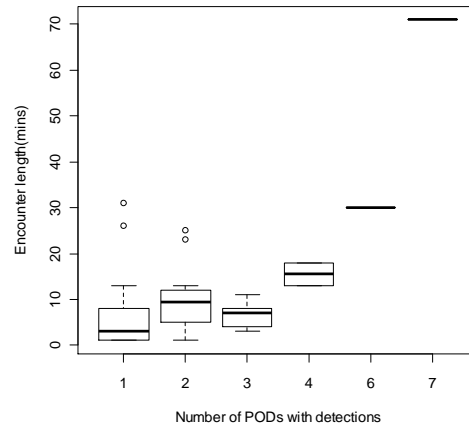
Figure 30: Box plot of lengths of encounters by the number of PODs that detected DPM in that encounter.

These data indicate that harbour porpoises are moving around the array during some encounters. However, because of the overlap in the predicted acoustic detection range of PODs in the array, it is difficult to tease out where an echolocating animal might be in relation to the array. For example, both PODs within a treatment could theoretically pick up an animal, depending on its orientation, at the maximum distance of 300m from the array. As a proxy for movement within encounters, the position of the POD that logged the first and last DPM within an encounter were collated and assigned a rank (1-8, dependent on position in the array). This rank relates to the distance between the last and first POD that recorded a DPM in that encounter. Figure 31 shows an example of distance rankings from POD 237. A rank of one would indicate that an encounter had ended and started on POD 237, while a rank of 4 would indicate that the encounter had ended on the southern POD of either of the neighbouring treatments.
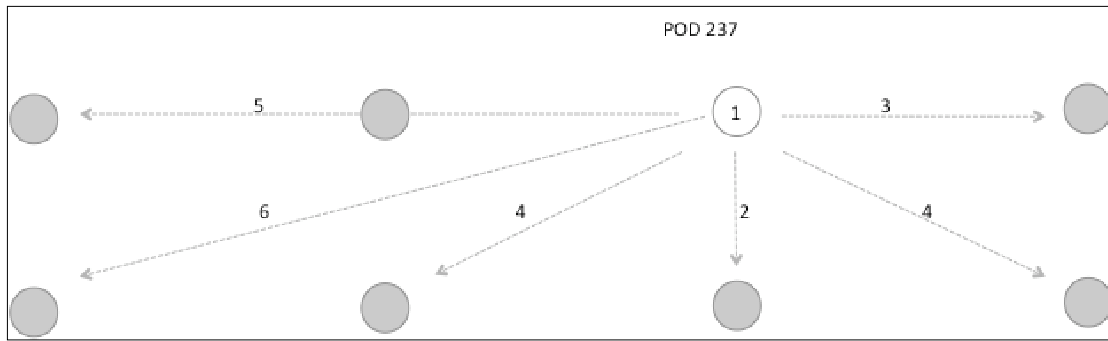
Figure 31: Example of distance ranking from a single POD in the array. Not to scale.

70% of all encounters began and ended on the same POD (n=53). Figure 32 shows the frequency of encounters per distance rank and Figure 33 the length of encounter for each distance rank.
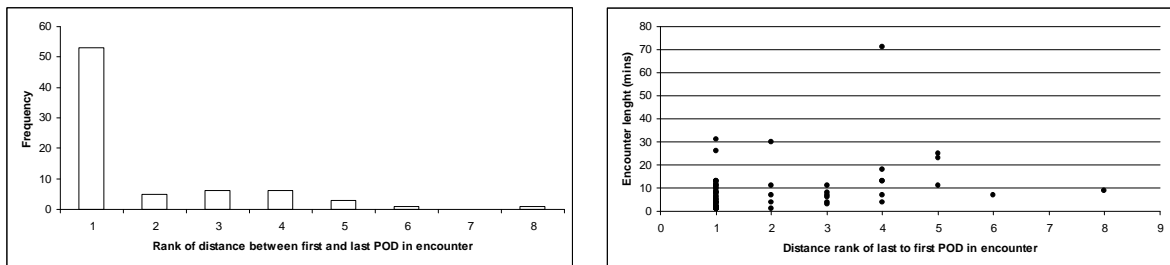
Figure 32: frequency of encounters by distance rank.   Figure 33: length of encounter by distance rank.

This shows that within an encounter lasting 9 minutes, the last harbour porpoise click train was detected by a POD at the furthest distance from the POD that first detected a click in that encounter. However, during this encounter no other PODs in the array logged porpoise detections, therefore this may either have been an individual that circumnavigated the array or may have been two separate individuals.

5.4.5   Investigating harbour porpoise echolocation activity in a commercial fishery.

5.4.5.1   Daily echolocation encounter rate

Data provided by the CWT were analysed to investigate harbour porpoise echolocation activity in a commercial fishery. Seven POD data files were analysed

which had been collected by four different boats during the CWT study. A total of 458 encounters were detected from the 7 POD deployment periods. Daily echolocation encounter rates were calculated. Table 16 provides mean encounter rate per day of deployment and standard errors. The overall average encounter rate per day was 0.9, although variability was high both between boats, and within net deployments within boats.

File ID	Mean enc. rate per hour	S.E
A	0.15	0.05
B	0.15	0.05
C	0.24	0.03
D	0.06	0.01
E	0.19	0.03
F	0.04	0.01
G	0.04	0.01

Table 16: Summary of encounter rates per hour by file.

A large proportion of days had no harbour porpoise click train detections. No encounters were recorded on 40% of days that PODs were deployed, and the proportion of days without encounters for individual files ranged from 0.03 to 0.61. Daily encounter rates were non-normally distributed with the exception of File E (Lilliefors test for normality: n=20, p=0.7) and linear regression analysis of all files (excluding file E) did not reveal any trend in encounter rate with day (r^2 =-0.017, p=0.42).

5.4.5.2 Encounter rate with diel phase

Encounter rates per diel phase were calculated for all POD data together. Mean encounter rate was highest during the morning and evening (Table 17).

	Phase length	S.E	Mean encounter rate	S.E
morn	1.21	0.02	0.20	0.04
day	12.22	0.13	0.14	0.02
eve	1.21	0.02	0.21	0.04
night	9.39	0.15	0.09	0.01

Table17: Summary of phase length and mean encounter rate by diel phase.

Echolocation encounter rates per diel phase were not normally distributed (Lilliefors test for normality, n=465, p <0.005) or homogenous (Bartlett's K-squared = 287.156,

df = 3, p-value < 0.005). Results of general linear hypothesis testing using Tukey's all pair wise comparisons showed that there were significantly more encounters during the evening than the night (p<0.01). This contrasts with the finding from the Bridlington bay trial where significantly more encounter per hour were recorded during the day.

5.4.5.3   Encounter rate with tidal phase

Echolocation encounter rates per tidal phase were not normally distributed (Lilliefors test for normality, n=1,653, p <0.005) or homogenous (Bartlett's K-squared = 49.6683, df = 9, p-value < 0.005). Results of general linear hypothesis testing using Tukey's all pair wise comparisons detected no significant effect of tidal state on echolocation encounter rate (all pair wise comparisons; p>0.5).

5.4.5.4   Encounter length and intensity.

Figure 34 shows a histogram of encounter lengths for all deployments combined. 84% of encounters were 1 minute long. Mean encounter length was 3 minutes (S.E. 0.16) and maximum encounter length was 28 minutes.
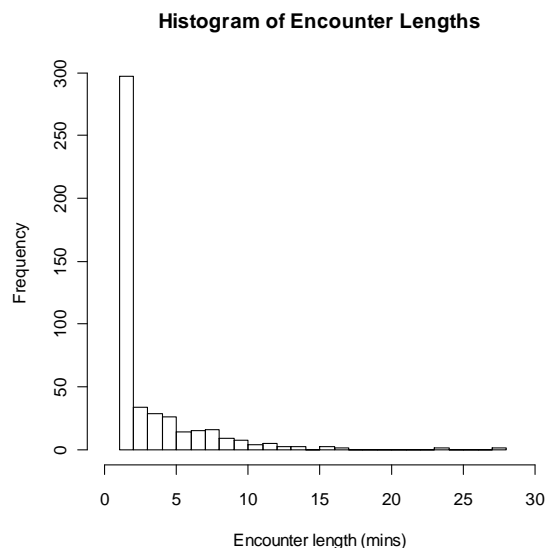
Figure 34: histogram of encounter lengths from all

POD data combined.

Table 18 provides a summary of the mean encounter length recorded during each POD deployment

File ID	Boat	Mean length (mins)	S.E
A	1	2.53	0.60
B	1	3.09	0.39
C	2	2.74	0.22
D	2	3.05	0.38
E	3	3.79	0.45
F	4	2.73	0.99
G	4	2.63	0.72

Table 18: Summary of mean encounter lengths by deployment.

Figure 35 shows the mean encounter length for each haul in each file. There is high variability in mean encounter lengths both within and between deployments. Points without error bars are those where there was only one encounter within that deployment.
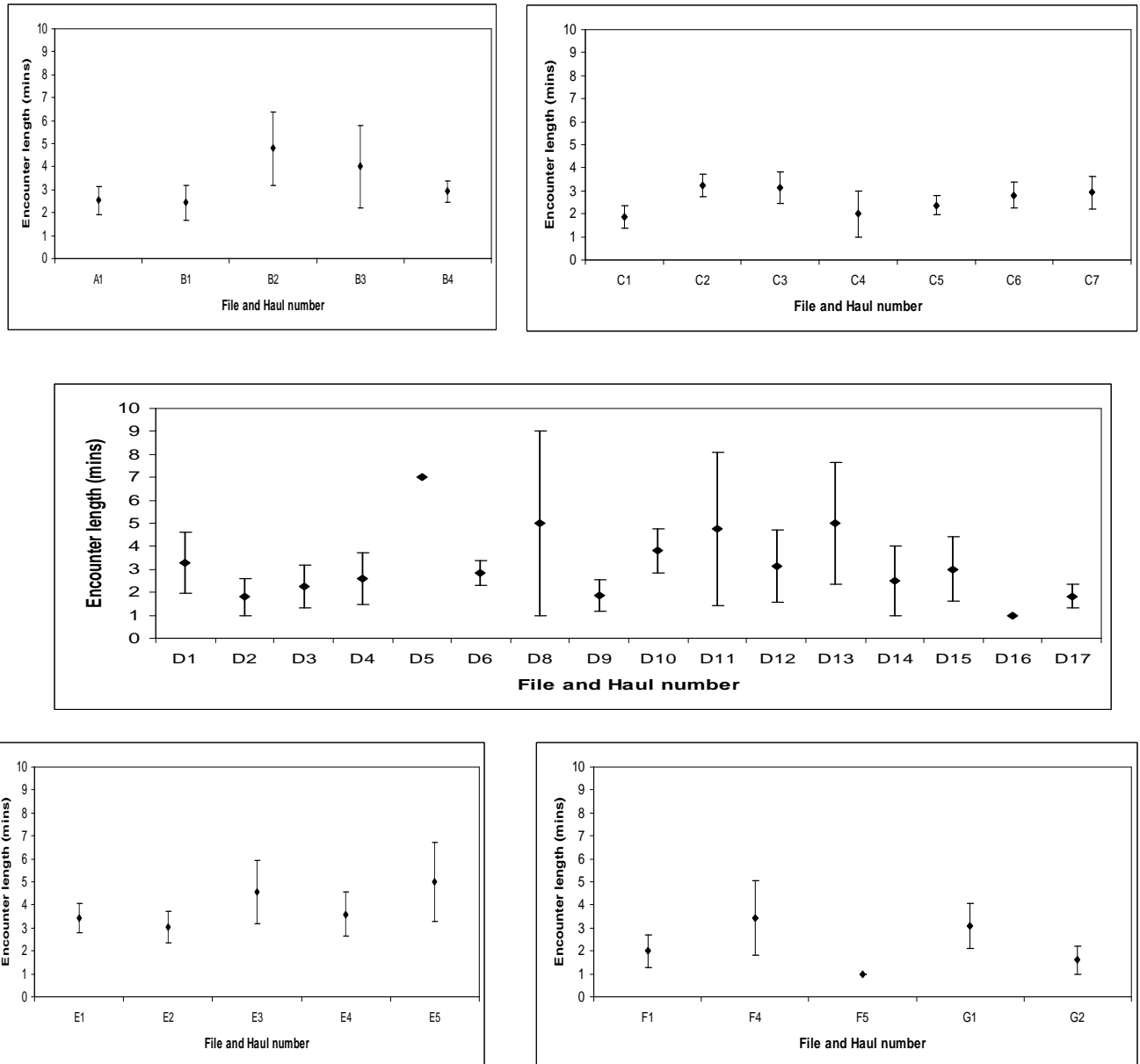
Figure 35: Mean encounter rate per haul for each POD file.

Figure 36 shows the proportion of minutes within an encounter in which at least 1 porpoise echolocation was detected for each file. Encounters that were 1 minute in length were removed prior to plotting as these have a proportion of DPM to encounter length of 1.
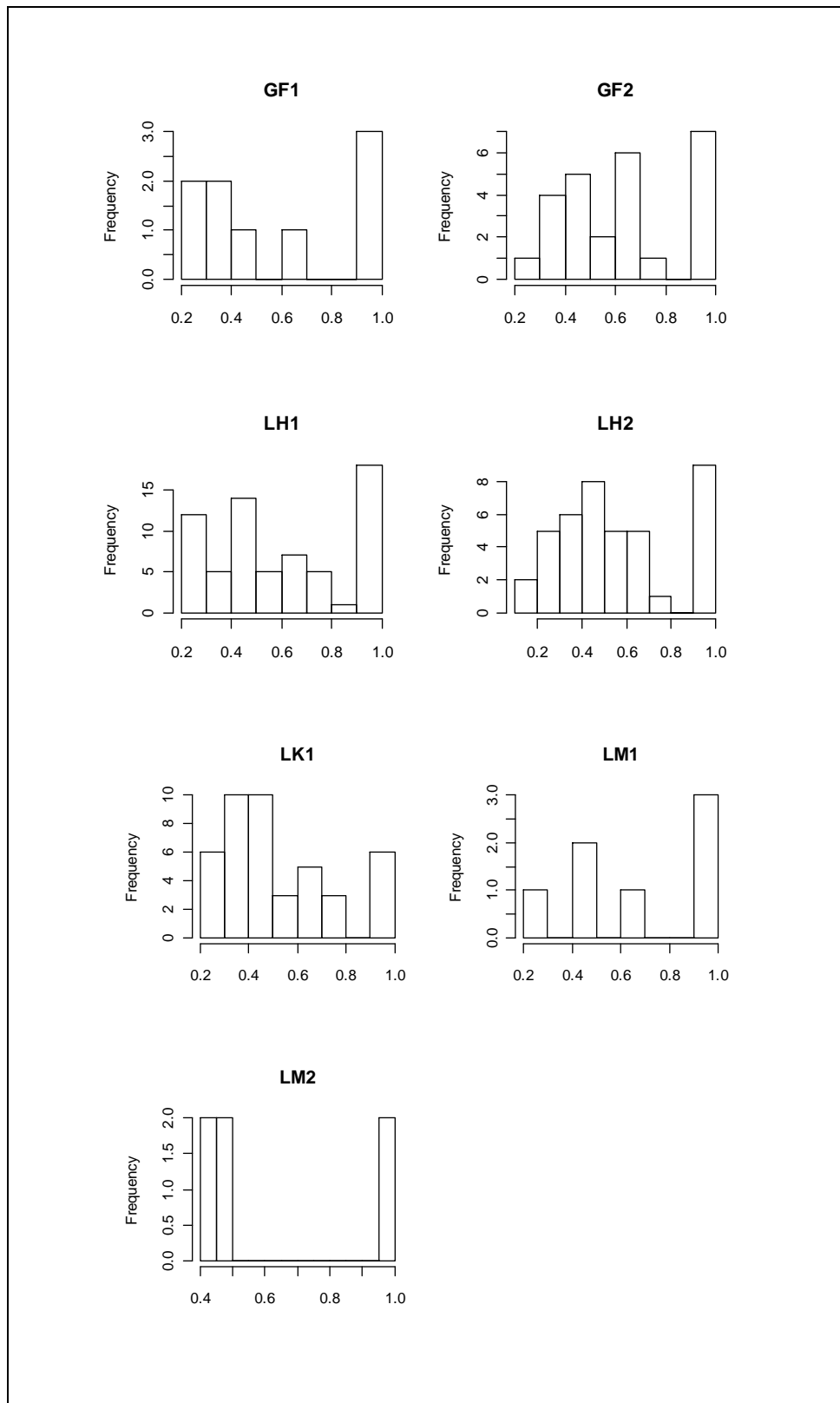
Figure 36: the proportion of DPM relative to encounter length.

5.4.5.5   Echolocation click trains

A total of 1,613 individual echolocation click trains were recorded by all PODs combined. Of these 4% fell into the category of fast click trains defined as having a minimum ICI <10ms. The minimum ICI recorded by a POD during the study was 1.025ms and the maximum was 363.61ms. Table 19 provides a summary of mean ICIs and the percent of ICIs in that phase which were less than 10 ms while Figure 37 shows the distribution of minimum ICIs per diel phase.

Phase	N	Mean ICI (ms)	% trains mICI <10ms
Morning	111	52	17.1%
Day	741	47	20.4%
Evening	137	35	20.4%
Night	624	32	25.3%

Table 19: summary of ICIs by diel phase.

Minimum ICIs were not normally distributed (Lilliefors n=1,820, p < 0.001) and heterocedastic (Bartlett's K-squared = 24.8847, df = 3, p-value <0.001).
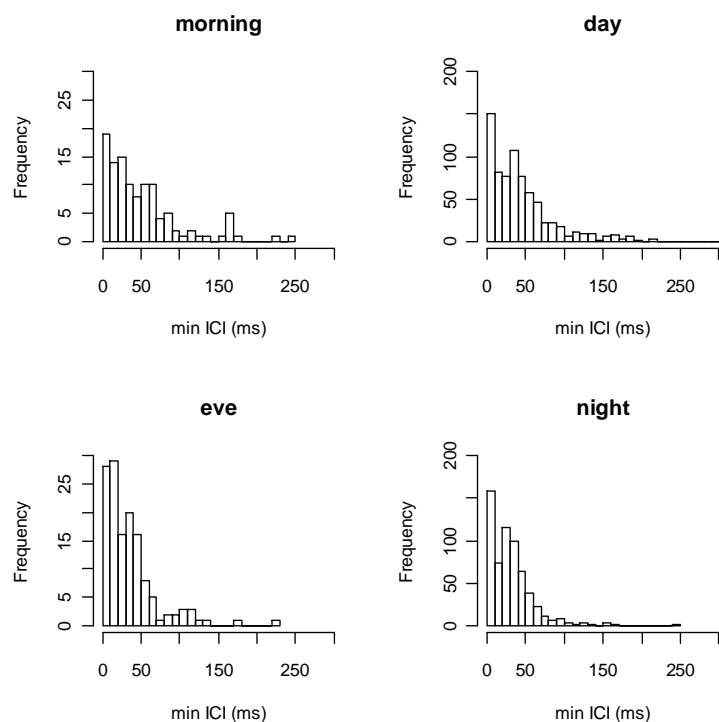
Figure 37: Distribution of mICI by diel phase.

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Results of general linear hypothesis testing using Tukey's all pairwise comparisons showed that mICIs were significantly lower in the night compared to the day (p<0.001) or morning (p<0.001), and significantly lower in the evening compared to the day (p<0.01) or morning (p<0.05). Therefore harbour porpoises are producing a higher proportion of fast click trains in the evening and nighttime than during the morning and day.

5.4.5.6 Proportion of fast to low click trains.

The proportion of fast and slow trains per encounter were then calculated and the relationship between this and encounter length was investigated. 10% of encounters had an FBR >1. A GLM with binomial error distribution was constructed and results showed no significant relationship (p>0.5). This contrasted with results from Bridlington where a significant positive relationship was found between encounter length and the proportion of fast trains in an encounter.

5.5 Discussion

5.5.1 Harbour porpoise occurrence in the study areas.

The aims of this chapter were to investigate the occurrence and echolocation behaviour of harbour porpoises at two different study sites and to determine whether echolocation behaviour was effected by the presence or absence of static gillnets. Data were collected at an experimental field site in Bridlington Bay, North Yorkshire and data collected from a commercial gillnet fishery were provided by the Cornish Wildlife Trust.

Harbour porpoise echolocation clicks were recorded on 100% of days that PODs were deployed at the Bridlington study site. Mean encounter rates per hour recorded by POD 272 and 278, over a 51 day period, were 0.25 and 0.35 respectively. Encounters lasted between 1 and 44 minutes, with an average encounter length of 4.9 minutes. In contrast, data provided by CWT for the south west site showed that PODs did not detect harbour porpoise on 40% of the days that they were deployed. In addition the data showed much more variability with respect to mean encounter rates per hour, which ranged from 0.04 to 0.24. Additionally, 84% of recorded encounters in the southwest lasted 1 minute in length compared to 33% of all encounters recorded in Bridlington. The disparity in the distribution of encounter lengths may be a result of experimental design. PODs deployed within a treatment in Bridlington, were separated by 200m, and therefore, the probability of a echolocating porpoise being detected by both PODs in a pair is likely to increase the length of an encounter if an animal moves through the array. In contrast, PODs deployed by CWT were attached to one end of a tier of commercial nets that were likely up to 1 km long in lenght. If porpoises in the south west study also moved along nets then it is unlikely that an animal would be detected on the same POD within the ten minutes or more period of silence by which the end of an encounter is defined. Harbour porpoises were detected more frequently in the Bridlington experimental site than around fishing nets deployed in the south west of the UK. The proportion of days without harbour porpoise detections in the south west may have been a result of differences in both the area and seasons in which PODs were deployed on fishing nets during the CWT

study. However, these differences may also reflect differences in the densities of harbour porpoises in both areas.

POD data have previously been used to investigate the diel echolocation behaviour of harbour porpoises (Cox et al. 2001, Cox & Read 2004, Carlstrom 2005, Todd et al. 2009); however, these observed diel patterns have not been consistent between these studies. Both Carlstrom (2005) and Todd et al. (2009) reported higher echolocation detection rates at night on PODs deployed off the west coast of Scotland, and in the North Sea respectively. Cox et al. (2001) also reported higher detections at night, however, in another study conducted in the same area (Bay of Fundy, Canada) they reported lower detections at night compared to the day (Cox & Read 2004). Data collected in Bridlington also showed that significantly more encounters per hour were recorded during the day. Although the mean encounter rate per hour was also higher during the day than at night from the CWT dataset the difference was not significant.

However, the way in which PODs are deployed may affect the data that is collected. For example, Kyhn et al. (2006) deployed three TPODs, one below the other, at depths of approximately 2m, 8m and 15m below the surface in Great Belt, Denmark. Comparing their results to those of Cox et al. (2001) and Cox & Read (2004), the authors suggest that diel patterns in echolocation activity may be related to depth, and that echolocation activity is highest at the surface at night. Perhaps more relevantly they showed that the distance between a TPOD and a reflecting surface affects detection rates of echolocation clicks, as any reflected clicks that are logged are likely to be assigned as true echolocation trains by the TPOD.exe software. DeRuiter et al. (2010) also showed that transmission loss varies with depth and conclude that detection probabilities and the distance at which PODs can detect porpoises will be affected by depth.

Not withstanding how deployment depth may affect data recorded by PODs there may be a number of other reasons why diel variations in echolocation rates are not consistent between study sites. The most important of these is likely to be the underlying spatio-temporal habitat use of harbour porpoise in different areas, especially in relation to the temporal behaviour of prey items, such as diurnal vertical migrations in the water column. Data collected by TDR and satellite tags deployed on

free-ranging harbour porpoise have shown that animals dive throughout the day and night. While Westgate et al. (1995) reported an increase in diving activity of harbour porpoise between the afternoon and evening in the Bay of Fundy (n=7), Teilmann et al. (2007) found highest diving rates occurred within daylight hours in Danish waters. It is likely that the differences in timing of the peak number of dives observed in the two studies is likely related to the behaviour of different prey species in the two study areas.

In Bridlington minimum ICIs were lower in the day than at night but this difference was not significant. In contrast the south west minimum ICIs were significantly lower at night compared to the day. However, at both study sites the proportion of trains with a minimum ICI <10ms was highest at night. Both Carlstrom (2005) and Todd et al. (2009) also reported highest minimum ICIs at night. Carlstrom (2005) hypothesized that an increase in echolocation rate during darkness may be a behavioural response by porpoise to compensate for the loss of visual information, while Todd et al. (2009) suggested an increase in the click trains with minimum ICIs <10ms may be indicative of increased foraging due to a nocturnal increase in prey availability.

A further factor, which may drive diel variations, will be the influence of tidal state on harbour porpoise distribution. Results from both the Bridlington and CWT trial showed no relationship between echolocation encounter rate and tidal phase. However, other studies have linked peak occurrence of harbour porpoise to different tidal phases (Johnston et al. 2005, Pierpoint 2008, Embling et al. 2010).

It is also possible that the context in which a POD is deployed may affect the diel echolocation behaviour or harbour porpoise. The only other study which deployed PODs with static gillnets and investigated temporal patterns in echolocation rates (Cox & Read 2004) found the same diel trend as I observed in Bridlington, and to a lesser extent from the CWT data. SMRU (2001) deployed a pair of PODs above and below each other, at an estimated distance of 65m, on a few occasions during their trial. The POD at the surface recorded porpoise detections every time it was deployed, but these detections did not always correspond to those detected on the POD deployed

on the gillnet. Additionally, porpoise activity logged by the POD deployed on the gillnet was up to 65% higher than the POD at the surface in some deployments.

5.5.2 Harbour porpoise echolocation behaviour in the presence and absence of gillnets.

Previous studies using PODs have shown that harbour porpoise are in the vicinity of commercial static gillnets much more often than they become entangled (SMRU 2001, Cox & Read 2004). However, no studies have previously compared the echolocation behaviour of porpoises in the presence or absence of bottom set gillnets.

There was no significant difference in echolocation encounter rate, encounter length or encounter intensity recorded by PODs deployed with and without a net. However, the proportion of fast to slow trains was significantly higher when a net was deployed and there was a significant positive relationship between the proportion of fast trains in an encounter and encounter length. As previously stated, fast click trains have been shown to be used when an animal locks its sonar onto an object during navigation or produces an echolocation buzz immediately prior to prey capture. The higher proportion of fast trains when a net is present could be interpreted as porpoises adjusting their bio-sonar to investigate or navigate around the net, or that there is more foraging opportunity when a net is there.

To date, only one study using POD data has tried to investigate feeding behaviour of harbour porpoises. Todd et al. (2009) used a feeding buzz ratio (FBR) to infer foraging activity in harbour porpoise around North Sea offshore gas installations, although the relationship between an FBR>1 and harbour porpoise foraging has yet to be demonstrated experimentally. Using this methodology, only 3% of all encounters recorded by POD 278 and 272 in Bridlington contained a FBR >1. Of these only 1 was from an encounter logged when no net was deployed. In contrast, 10% of encounters from the CWT data had an FBR>1. However, there was no significant relationship between the proportion of fast trains in an encounter and encounter length from this data set, although, this may be due to the high proportion of encounters lasting only 1 minute that were recorded during this study.

This metric did not provide useful information into the possibility that harbour porpoises were foraging in the vicinity of static gillnets or were acoustically inspecting nets, therefore, the proportion of fast to slow click trains recorded in each encounter were examined. Whether a higher proportion of fast trains in the presence of gillnets represents the closer inspection of nets acoustically by harbour porpoise or is an indicator of foraging remains unclear. In a captive experiment Kastelien et al. (1995a) found that harbour porpoises navigating around ropes in a swimming pool generally used ICIs of 40ms or longer. In a subsequent study (Kastelein et al. 1995b) when nets were placed into the pool they found that in general the harbour porpoise did not direct click bursts at the net, and instead used similar click trains to those observed in the absence of nets. Almost all click bursts were recorded when live fish were introduced into the pool. These results suggest that at least for captive harbour porpoises, fast click trains occurred more frequently when animals were engaged in foraging than when they were navigating around nets. A study by Lauriano & Bruno (2007) which used version 3 TPODs to investigate the echolocation behaviour of bottlenose dolphins in the vicinity of traps and two types of trammel net (targeting lobster and red mullet respectively) found that dolphins produced statistically faster click trains around red mullet nets compared to the other two gear types. Bottlenose dolphins had previously been reported to depredate red mullet nets and although the authors stated that small sample size meant it was not possible to relate fast click trains recorded around these nets to depredation events, they concluded that such trains could be indicative of dolphins foraging. The results of these studies suggest that fast click trains recorded in association with nets could be produced when cetaceans are engaged in foraging.

Inferring the closest approach of harbour porpoises to gillnets.

Koschinski et al. (2006) used the distribution of minimum ICIs recorded by PODs deployed with a standard and an acoustically reflective net ($BaSO_4$) to infer the closest approach of harbour porpoises to each net treatment. They found a significant difference in the distribution of median ICIs between both treatments, and concluded that a higher median for the $BaSO_4$ net meant that porpoises were detecting that net at further distances than the standard net.

Data from an acoustic logger deployed directly on a free-ranging harbour porpoise showed the animal used an average ICI of 80.5ms (Akamatsu et al. 2007), but ranged from "a few" milliseconds up to 150ms. Villadsgaard et al. (2007) collected echolocation data of free ranging harbour porpoise using a vertical hydrophone array. The median ICI they recorded was 58ms (range: 30-200ms), however, they also recorded one click sequence with ICIs of 6ms but excluded these from their analysis.

Using the average ICIs recorded within a click train there was no significant difference in the distribution of clicks logged in the presence or absence of nets, or between times when the single float line net and net with polypropylene floats was deployed. Differences in median average minimum ICI varied from 1-9ms for net versus no net deployments, and from 4-7ms for single float line versus polypropylene float line deployments. Although there was no significant difference in the median average click rates logged between different treatments, they were always lowest on POD 278. This POD was placed in position 2 in the array, and with the exception of deployments 1-4, a net was always deployed in positions 1 and 3 on either side of it. Again there was no significant difference in the distribution of minimum ICIs recorded for either of the treatment pairs (net versus no net, single float line versus polypropylene float line). However, a pattern does exist between minimum ICI recorded within a click train and treatment type, and these were consistently lower in the presence of a net than when no net was present, and lower in the vicinity of the net with the polypropylene floats compared to the net with the single float line. The latter result may represent closer acoustic inspection of this more complex float line type by harbour porpoise.

Inferring the detection range of harbour porpoises to gillnets.

The target distance of an object from a click source can be calculated as the time it takes for a click to reach that object and time it takes for that echo to return. The time it takes for an odontocete to process the information contained in such a returning echo is termed the lag time. Lag times in harbour porpoise have been shown to range from 14 to 36 ms. Koschinski et al. (2006) calculated the target detection range of two different nets using lag times between 11.7 and 35ms. Following this methodology the target distance of each treatment to an echolocating harbour porpoise was calculated

using the medians average ICIs and minimum ICIs for all encounters recorded by PODs with that treatment (Table 17). Target distance calculated from average ICIs ranged from 14.3 - 26.3m when no net was present and from 15 – 19m when a net was present. These estimates fall into the range of 13-26m that Villadsgaard et al. (2007) recalculated for detection distances reported by Kastelien et al. (2000) using a higher harbour porpoise echolocation click source level of 191 dB re 1 µPa pp. Calculated target detection distances to the net with a single float line was 15m for both deployment periods and ranged from 9.8 –18m for the net with polypropylene floats.

	Target distance (m) using median of average. ICI		Target distance (m) using median of min. ICI	
Deployment	**No net**	**Net**	**No net**	**Net**
5 & 6	14.3	15.0	15.0	16.5
9 & 10	26.3	19.5	24.8	21.8
Deployment	**Floats**	**Single**	**Floats**	**Single**
1 - 4	9.8	15.0	7.5	12.8
7 & 8	18.0	15.0	15.0	18.8

Table 17: Estimated detection distances of different treatments using a lag time of 20 ms.

If the median ICI really does represent the average detection range of a porpoise to a net then these results raise a number of questions. We would expect that detection ranges would be greater when a net is present than when no net is present, but this is only the case for deployments 5 & 6 and is opposite for deployments 9 & 10. Likewise, we would expect that a float line with polypropylene floats would be detected at greater distances than a net with a single polypropylene float line. However, this is only the case in deployments 7 & 8. The only clear pattern is that the median of the average ICI for all trains is always highest at POD 272. Given there is no difference in detection ranges with or without a net it would suggest that animals are detecting the PODs possibly before the nets. This is not illogical given the target strength of a POD is –1.8dB and therefore provides a strong returning echo to an echolocating porpoise. The fact that POD 272 always recorded the highest median average ICIs could be due to between-POD variability, or to the position of the POD in the array.

As POD 272 was one of two PODs deployed on the outer eastern extremity of the array it is possible that the higher median average ICIs recorded by this POD represent the first detection of the array by an approaching harbour porpoise. Lower median average ICIs on POD 278 may then reflect the reduction of bio sonar range by harbour porpoise in an acoustically more complex environment (the array). Verfuss et al. (2005) hypothesize that porpoises lock onto specific places, or landmarks, in an environment and that the click interval they produce is relative to the distance to this landmark. In addition, they found that lag times were longer when captive animals were navigating through a more complex experimental set up and concluded that this may be due to animals needing more time to process more complex information from returning echoes. Estimates of target detection distance are very sensitive to the lag time used, and it is possible that animals within the array could use longer lag times as they are in a more complex situation. Therefore, it is unlikely that the detection ranges calculated above actually reflect the true distance at which harbour porpoise detected different treatments in the array. However, as the same lag time was used for all treatments it is clear that detection distances were not significantly greater when a net was present to when no net was present.

5.5.3   Harbour porpoise movement around gill nets.

Due to CPOD failures, data collected simultaneously by all PODs in the array were only available for deployment 2. There was temporal and spatial variability in harbour porpoise detection when data were analyzed by individual POD, pairs of PODs and for all PODs combined. The average encounter, calculated by combining data collected by all PODs in the array, lasted 7.8 minutes but encounter lengths ranged from 1 to 72 minutes in length. There was a general trend between encounter length, and the number of PODs that recorded DPMs within an encounter. This suggests that the longer harbour porpoises stay in the vicinity of the array, the more they move between treatments. Unfortunately, due to the overlap in the predictive acoustic detection range of PODs in the array it was not possible to track porpoise movement around treatments. However, when the data were ranked by POD position (relative to the first and last DPM in an encounter) 70% of encounters began and ended at the same POD. The lengths of these encounters ranged from 1 to 31 minutes.

Interestingly, in 1 encounter, which lasted 9 minutes, the last harbour porpoise click train was detected by a POD at the furthest distance from the POD that first detected a click in that encounter. During this encounter no other PODs in the array logged porpoise detections. Therefore, this may either have been one animal that circumnavigated the array, or there may have been more than one porpoise in the array concurrently. There were two instances when a DPM was recorded simultaneously on PODs from three different treatments. For this to be possible more than one animal must have been present in the array at that time. This is an additional source of variability when using POD data to infer echolocation behaviour, because analyses conducted in this chapter are based on the assumption that an encounter represents a single animal being present in the array. This is clearly not the case in some instances, but it is difficult to ascertain from POD data the number of animals that are echolocating, unless, during visual inspection of the data you observe two different click trains overlapping. However, even then, it is not currently possible to assign preceding or subsequent trains to one or other of the animals.

5.5.4 The effect of the presence of a POD on echolocation behaviour.

Given the high target strength of the POD (-1.8dB), relative to the target strength of static gillnets, it is highly likely that the echolocation data collected, both in the presence and absence of nets, includes click trains which were generated during inspection of the PODs by harbour porpoise. Rising click rates in sequences of trains have been attributed to harbour porpoise adapting their bio sonar to account for the two-way travel time of an echolocation click as they approach a landmark (Tregneza *pers comm.*). These sequences have been termed landmark sequences and can be extracted from the CPOD.exe software. Although the CPOD.exe algorithm did not detect any landmark sequences in the Bridlington data, a number of probable sequences were identified after visual inspection of the data. Figure 38 shows two screen grabs of such landmark sequences taken from POD 272 and 278 recorded at times when *neither* POD was deployed with a net.

Time (seconds)

Time (seconds)
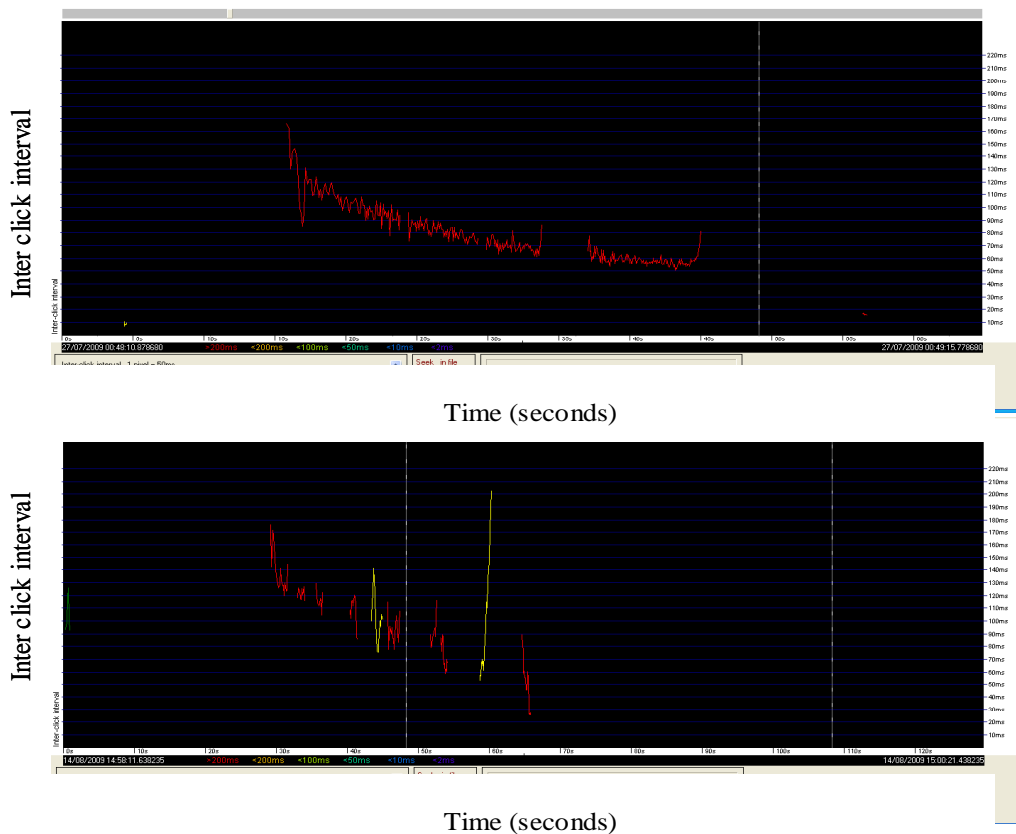
Figure 38: Screen grabs of "landmark sequences" on POD 278 (top) & 272 when not deployed with a gillnet.

These data suggest that harbour porpoise do investigate PODs with their bio sonar and means that the data collected on echolocation behaviour in the presence and absence of nets will be confounded by changes in echolocation behaviour of porpoise in the presence of a POD. The amount of time an animal investigates a POD relative to a net if it is there, will likely affect comparisons of calculated encounter length and encounter intensity. The fact that there was no significant difference in either of these metrics in the presence or absence of a net may therefore be due to the inherent affect of a POD on harbour porpoise echolocation behaviour. However, it remains clear from the results of this chapter that harbour porpoise do use a significantly higher proportion of fast click trains in the vicinity of a net compared to when a POD is deployed without a net.

Since it is clear that PODs make a very strong acoustic target to echolocating harbour porpoise it is possible that when deployed with nets they may alert animals to the presence of the net and make them acoustically inspect in more detail the area around the POD. Akamatsu et al. (2007) found that a free ranging harbour porpoise did not

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swim for more than 10m without producing intense echolocation clicks and that when the animal swam without echolocating, the distance it traveled was generally shorter than the estimated sonar range calculated from the last clicks it produced. Periods when the animal did not echolocate lasted for more than 1 minute, however, it is possible that in these times the animal was in fact echolocating but may have produced low intensity clicks which the Atag would not have recorded. Given the higher probability of a porpoise detecting a POD than a gillnet float line it is possible that having detected the PODs, porpoise then inspected the area, including the deployed gillnets in more detail.

If animals do acoustically inspect the area around a POD in more detail then it is possible that deploying PODs on nets (or more suitable passive reflectors) may reduce harbour porpoise bycatch. Results from early trials using passive reflectors to increase the acoustic reflectivity of nets to echolocating cetaceans were ambiguous and since then acoustic modification of gillnets has focused on increasing the target strength of the mesh panel. However, results of a trial testing bycatch rates of Franciscana dolphins in acoustically reflective nets showed these nets did not reduce the bycatch rates of this species (Chatper 3). Data collected by SMRU (unpublished) have recorded two incidences of harbour porpoise bycatch in a gillnet with a POD deployed. However, in one of these hauls the gillnet had not set properly and was floating at the surface. No bycatch was recorded in 36 observed gillnet hauls with PODs attached in the south west of England (SMRU et al. 2001), or in 25 hauls in the Bay of Fundy (Cox et al. 2004). However, it is clear that these samples sizes are too low to test the affect on bycatch rate. Additionally, during the CWT trial, 1 harbour porpoise was caught in a net with a POD deployed, however, information on where this animals was caught in relation to the POD was not available.

5.5.5 Conclusions

The main aim of this chapter was to investigate whether or not harbour porpoise are attracted to static gillnets. Alhough there was no significant difference in encounter rates, encounter length, or encounter intensity recorded by PODs deployed with or without a net, it is clear from the occurrence of landmark sequences on PODs deployed without nets, that the presence of a POD itself affects the echolocation behaviour of porpoises.

Only limited data were available to investigate the movement of harbour porpoise around gillnets. Results show that porpoises moved between treatments in the array but also circumnavigated the array. Over 50% of encounters were recorded by both PODs deployed within a treatment, suggesting that for a high proportion of time animals do indeed move along nets when they are present.

The most interesting result is the significant difference in the proportion of fast echolocation clicks produced by porpoise when they are in the vicinity of a net compared to when no net is present. But it is impossible to say whether these faster click trains are related to closer acoustic inspection of the net or are indicative of foraging behaviour. Only a small proportion of encounters contained a feeding buzz ratio that was greater than 1 in Bridlington (3%), while the proportion in recorded in the south west was slightly higher (10%). However, calculating feeding buzz ratio using all trains in an encounter may produce an underestimate of the true ratio, as depending on encounter length, a higher number of navigational trains may be recorded. The effect of float line type on echolocation activity was more ambiguous.

These results support previous observations that harbour porpoise are in the vicinity of nets more often than they become entangled. They also suggest that porpoises may be foraging around nets. Kindt-Larsen (2007) analyzed the stomach contents of bycaught harbour porpoise and hake captured during the same haul in a commercial static gillnet fishery in Denmark. Though Kindt-Larsen found no significant overlap in prey items in the stomachs of porpoise or hake, sample sizes in this study were small. Clearly, further evidence will be needed to confirm that harbour porpoise are actively foraging around gillnets. The risk of entanglement to a foraging individual may be higher under a number of scenarios. The TS of the prey item it is approaching may mask echoes from the net mesh and therefore the porpoise may not detect the net, resulting in entanglement. Or, the porpoise may not concentrate on the closeness of a net in the final moments of prey pursuit. Harbour porpoise have been observed to forage by "bottom grubbing". During this foraging behaviour an animal positions its rostrum close to the seabed, focusing its echolocation clicks downwards (Stenback 2006). It is clear that animals engaged in such behaviour would have a lower likelihood of detecting a bottom set gillnet before entanglement would occur, or during foraging may get closer than intended to a net it has previously detected.

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6. Chapter 6: An assessment of the behaviour of bottlenose dolphins interacting with a bottom trawl fishery.

6.1 Abstract

Analysis of underwater video footage recorded inside trawl nets in the Pilbara bottom trawl fishery, Australia, were examined to determine the occurrence and behaviour of bottlenose dolphins (*Tursiops sp.*) interacting with this fishery. Bottlenose dolphins were present inside trawl nets more frequently than they were bycaught and were often actively foraging inside the nets. The deployment of excluder devices have significantly reduced dolphin bycatch rates in this fishery, however, it was unclear whether part of this observed reduction was due to bycaught animals falling out the of the escape hole during haul back. Using a Bayesian approach a posterior distribution of the probability of a dolphin being caught but not landed on deck was calculated. Results showed that when these possible unaccounted bycatches are considered there remained a probability of 0.62 that the reduction in bycatch in nets with an excluder grid was true. Video analysis indicated that dolphins freely swimming inside trawl nets were most frequently orientated towards the vessel, indicating that changes to the current design of the excluder grid could be made to further reduce bycatch rates in this fishery.

6.2 Introduction

At least 25 species of cetaceans and 36 species of pinnipeds (Fertl and Leatherwood 1997, Northridge and Hofman 1999, Hamer and Goldsworthy 2006) have been reported as bycatch in trawl fisheries around the world. The likelihood that marine mammals will interact with a particular trawl fishery may be increased by a number of factors. A temporal or spatial overlap between a trawl fishery and marine mammal species may exist if the distribution of both is related to larger environmental factors such as biological productivity. For example, if the target species of the fishery and prey species of the marine mammal are the same, or concurrent, the likelihood that the distributions of the marine mammal species and fishery will overlap will be increased. Fertl and Leatherwood (1997) suggested that trawl nets may present easy access to a concentration of prey items which are less energetically costly for foraging cetaceans to exploit, or may present an opportunity for animals to forage on prey species which

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are usually inaccessible. If marine mammals are motivated to interact with trawl fisheries in order to increase their foraging success, then it is possible that such interactions will result in a higher probability of bycatch occurring.

Analyses of the stomach contents of marine mammals bycaught in a number of different trawl fisheries has shown an overlap between ingested species and those targeted by the fishery (Waring et al. 1990, Couperus 1997, Tilzey et al. 2006). However, without direct observations of marine mammals feeding inside trawls such results may simply reflect that animals were feeding on the same species targeted by the fishery but not necessarily utilizing the nets to do so. In some trawl fisheries, where marine mammal bycatch has been observed, there is no overlap in target catch and prey species (Crespo et al. 1997, Northridge et al. 2004), indicating that animals may be foraging on non-target species associated with the fishing operation. However, not every interaction between marine mammals and trawl nets results in bycatch; in many instances where associations between animals and nets are common, bycatch rates remain low (Chilvers and Corkeron 2001). For cetaceans, the species most often documented feeding in association with trawls is the bottlenose dolphin *(Tursiops spp.),*(Fertl and Leatherwood 1997, Broadhurst 1998, Chilvers and Corkeron 2001, Chilvers et al. 2003, Pace et al. 2003, Svane 2005, Fortuna 2006, Gonzalvo et al. 2008, Anon. 2009, Scheinin 2010). Reports of feeding associations include observations of dolphins foraging behind trawlers (Fertl and Leatherwood 1997, Fortuna 2006), feeding on fish discarded by trawlers (Chilvers et al. 2003, Pace et al. 2003) and direct observations of dolphins taking fish underwater from the ends of shrimp trawls (Broadhurst 1998). With the exception of the underwater footage obtained by Broadhurst (1998), cetacean-trawl foraging interactions are usually inferred by observations of the surface behaviour of cetaceans in the proximity of trawl nets.

The operational characteristics of trawl fisheries, combined with low marine mammal interaction rates, makes it difficult to obtain underwater observations of animals directly feeding or interacting with trawl nets (Northridge et al. 2004). To date the majority of underwater observations have been of pinnipeds (Shaughnessy and Davenport 1996, Browne et al. 2005, Hamer and Goldsworthy 2006, Lyle and Willcox 2008) with only limited underwater footage of cetaceans (Stephenson and

Chidlow 2003, Northridge et al. 2005, Stephenson and Wells 2006). One incidental observation of a cetacean species seen to enter the mouth of trawl nets is described in Waring et al. (1990) who observed pilot whales feeding around, and in, the opening of nets in the Atlantic mackerel trawl fishery. Marine mammals that are motivated to enter trawl nets to forage will have an increased probability of becoming captured, and developing mitigation strategies to reduce the likelihood of such entanglements occurring will be specific to the species and fishery in question.

Although a substantial amount of effort has been spent on the development and testing of marine mammal bycatch mitigation strategies, especially in Europe, Australia, New Zealand and the USA, the mechanisms by which individual cetaceans become entangled in fishing gear remain poorly understood (Werner et al. 2006). However, a number of studies have focused on reducing marine mammal bycatch in trawl fisheries through technical measures (de Haan et al. 1998, Browne et al. 2005, Northridge and Mackay 2005, Chilvers 2008, Lyle and Willcox 2008). In general, these studies have utilized one of two mitigation technologies: acoustic deterrent devices (Northridge et al. 2004, Stephenson and Wells 2006) or excluder devices (Gibson and Isaksen 1998, Northridge et al. 2003, Browne et al. 2005, Northridge et al. 2005, Tilzey et al. 2006, Lyle and Willcox 2008). The efficacy of acoustic deterrent devices to reduce cetacean bycatch remains unclear (Stephenson and Wells 2006) and to date only ongoing trials in the UK and French pelagic bass pair trawl fishery have shown promising results (ICES 2009). Studies to assess the performance of excluder devices at mitigating marine mammal bycatch have also had mixed results (Northridge and Mackay 2005, Lyle and Willcox 2008) and as with the testing of any bycatch mitigation device, definitive results are often difficult to obtain. This is because bycatch events are generally rare and therefore a high proportion of the fishery, if not the entire fishery, must be observed, which is very costly, to ensure an appropriate sample size to detect statistically significant differences between tows with or without mitigation devices. Finally, compounding these issues further is the fact that, unless the mechanism by which bycatch reduction occurs is known, implemented devices may give inconsistent results between seasons (Northridge et al. 2004). Therefore, in order to assess the performance of excluder grids at reducing bycatch, a way to measure or predict escape and survival rates of marine mammals interacting with such devices is needed.

6.2.1 The Pilbara Finfish Trawl Interim Managed Fishery (PFTIMF)

The Pilbara Finfish Trawl Fishery is an interim managed fishery[1] which operates between 114^o10'E and 120^oE off Western Australia and is restricted to an area of 15,000 square miles which is subdivided into 6 management areas (Fig. 1). The fishery has operated since the 1970's when it was predominantly prosecuted by Taiwanese boats. Domestic trawlers entered the fishery in the mid-1980's, and since 1990, the fishery has been closed to foreign vessels.

The fishery targets a number of finfish species including red emperor (*Lutjanus sebae*), spangled emperor (*Lethrinus nebulosus*), saddletail snapper (*Lutjanus malabaricus*), goldband snapper (*Pristipomoides multidens*), frypan snapper (*Argyrops spinifer*), crimson snapper (*Lutjanus erythroptesus*), Rankin cod (*Epinephelus multinotatus*), rosy threadfin bream (*Nemipterus furcosus*), and several flatfish species. Vessels in the fishery tow a single bottom trawl net of 200 –300 mm mesh size on the wings and 100 mm mesh in the codend. Gear restrictions exist under the current management plan. These state that the total net length including the sweeps (90m), bridles (25m) and head ropes (36.58m max.) must not exceed 274.32m. Bobbins with a maximum diameter of 350mm are placed at approximately 0.3m intervals along the footrope. Tows last 3 hours on average but range between 30 minutes and 5 hours in length, and occur in depths between 50 and 100 metres.

The first management plan was put in place in 1998 and the fishery is managed using effort limitation, area closures and gear restrictions (Stephenson & Chidlow, 2003). There are 11 Managed Fishery Licences (MFLs) to operate in the fishery, however due to individual transfer of effort which is allowed under the management scheme only 4.3 full time vessels (with approximately 225 days of effort per vessel per year) operate in the fishery. Since the implementation of the management plan, total effort in the fishery has been reduced and area closures implemented, including closing zone 1 and Area 6 to trawlers. Both effort reductions and area closures have predominantly

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[1] "Interim Managed Fisheries" are declared under the *Fish Resources Management Act 1994* and are governed through a management plan which can stipulate that the plan only has effect for a specified period. In the Pilbara Trawl Fishery the management system is based on individual transferable effort (ITE) units.

been aimed at maximising yield whilst maintaining sustainable stock levels of indicator species such as red emperor and Rankin cod.
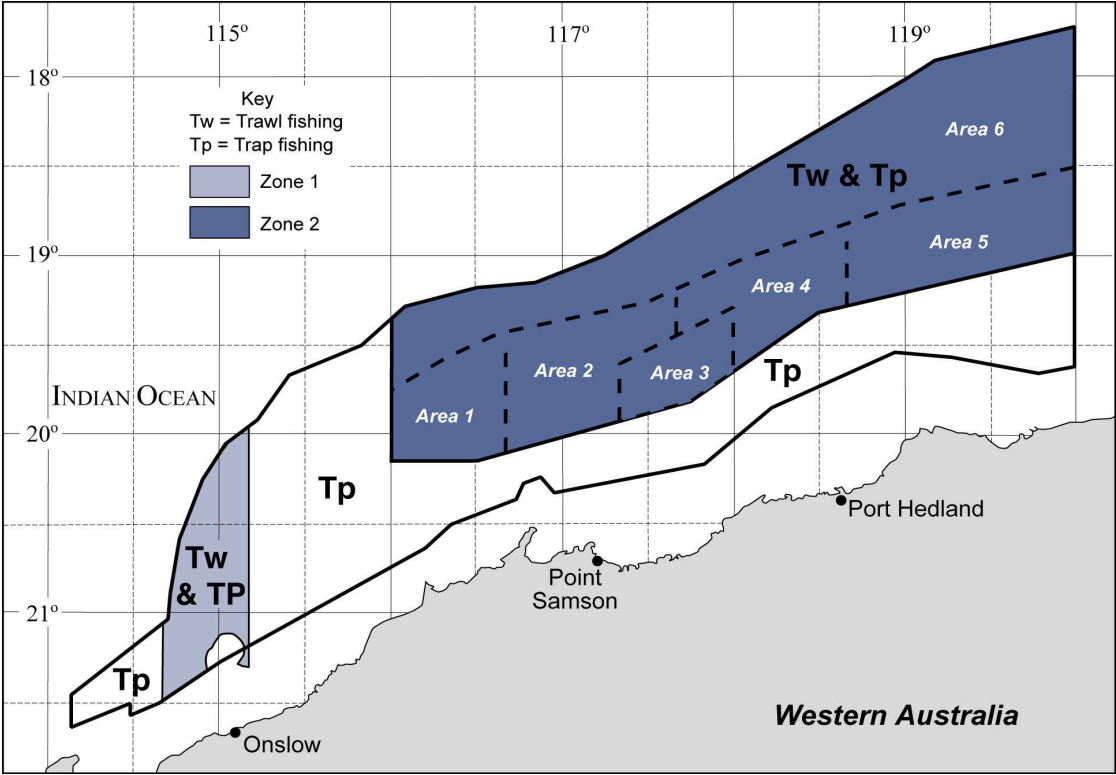
Figure 1: Areas 1 to 6 refer to the management regions in Zone 2 of the trawl fishery. Zone 1 has been closed to trawling since 1998. Taken from Stephenson & Wells (2006)

The fishery is allocated approximately 21,000 hours of effort (~ 5000 tows), which is spread across 4 of the 6 management areas. Table 1 shows the allocation of effort per management area in 2007.

	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Total Effort
Effort in Hours	9,596	3,797	0	3,528	4,627	0	21,548

Table 1: Allocation of fishing effort by management area in the PFTIMF in 2007.

The fishery operates throughout the year although there is a reduction in effort during months of bad weather (December to March) and during April and May when some vessels focus on prawn fishing.

6.2.2   Bycatch reduction of protected species in the Pilbara trawl fishery

The fishery incidentally catches a number of species protected under Australian and or international law. These include bottlenose dolphins, turtle species, sawfish, pipefish, sea snakes, sea horses and sea dragons. Bottlenose dolphins are listed as a protected species in Australia under section 248 of the Environment Protection and Biodiversity Act (EPBC*)* Act 1999.

In 2002 the Natural Heritage Trust funded the first inventory of bycaught species in the fishery, and resulted in the first bycatch estimate of 64 dolphins a year (Stephenson and Chidlow, 2002). Underwater video was recorded during this project on a number of tows and resulted in 50 minutes of edited footage, which provided the first indication of the way in which bottlenose dolphins were interacting with the trawl net. Dolphins were observed foraging both inside and outside of the trawl net, with individual dolphins observed drifting backwards (head orientated towards the vessel) into the net and then holding position to feed on incoming fish. This video provided the first indication that dolphins were aware of the net, they could manoeuvre easily inside the net and were specifically entering the net to forage.

In 2004/2005 the Fisheries Research and Development Corporation (FRDC) funded a project to test the effectiveness of pingers and exclusion grids with the aim of reducing dolphin bycatch in the fishery. Video footage collected from 14 of these tows was deemed to be of sufficient quality to count the number of dolphins recorded on screen during tows with or without pingers deployed. There was no significant difference in the number of dolphins counted on screen between tows with or without pingers (Stephenson, 2006a). In contrast to the acoustic pingers, exclusion grids, which were also trialled during this project, showed some success in reducing dolphin bycatch and so evaluation of exclusion grids in this fishery continued in 2006 with a Department of Fisheries Development and Better Interest Fee (DBIF) funded project.

This project coupled underwater video footage with grid deployments to assess the effectiveness of exclusion grids at reducing the bycatch of dolphins and other protected species including turtles, sharks, and rays (Stephenson 2006b). Exclusion grids became mandatory on the 1st of March 2006, shortly after the commencement of the DBIF project. Between December 2005 and the 31st of July 2006, 1,384 tows were

observed, through a combined industry and Department of Fisheries funded observer program and the DBIF funded grid trial project. An underwater video camera was deployed on 446 of these observed tows. Figure 2 shows a picture of one of the grids deployed in situ in a trawl net.
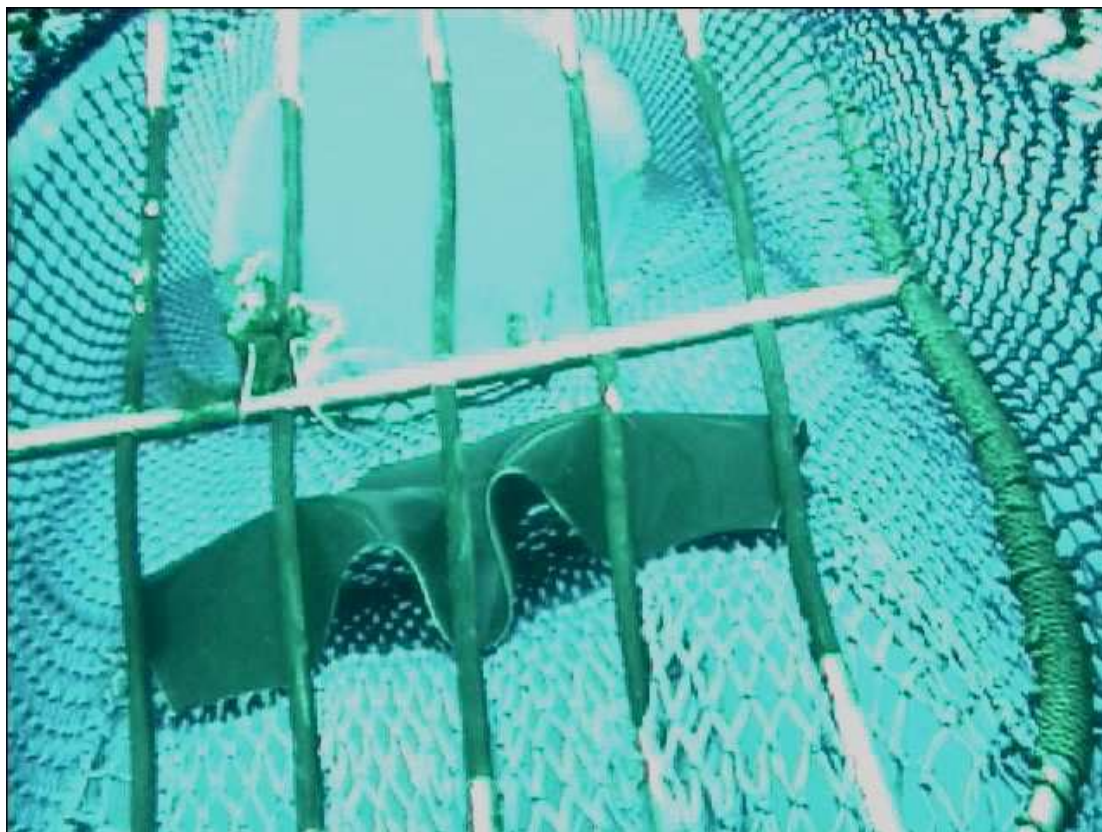
Figure 2: Exclusion grid in-situ in a trawl net. Taken from Stephenson & Wells, 2006.

During this project, the estimated dolphin bycatch rate was reduced from 15.2 dolphins per 1000 tows without grids, in 2005, to 7.8 dolphins per 1000 tows with grids deployed, in 2006. Because of this observed reduction in dolphin bycatch the Department of Fisheries continued the industry funded observer program to August 2007. Results from 2007 showed a slight, but non-significant, increase in dolphin bycatch with an estimate of 10 dolphins caught in every 1000 tows. While the use of exclusion grids have been shown to significantly reduce bottlenose dolphin bycatch in this fishery, the mechanism leading to this reduction is less apparent. In particular there were concerns that the observed reduction in bycatch rates may have been a result of dolphins being caught in the net but subsequently being expelled through the

escape hole either prior to or during haul back and therefore leading true bycatch rates to be underestimated.

An 18-month FRDC funded project commenced in July 2008 with the aim of improving the design of exclusion grids in order to further reduce dolphin bycatch in this fishery. This included moving grids further forward in the net towards the vessel.

All video data analysed for this chapter were collected from nets where exclusion grids were deployed at the beginning of the codend extension at the 100mm meshes. Figure 3 provides a schematic of trawl net dimensions and positioning of excluder grids.
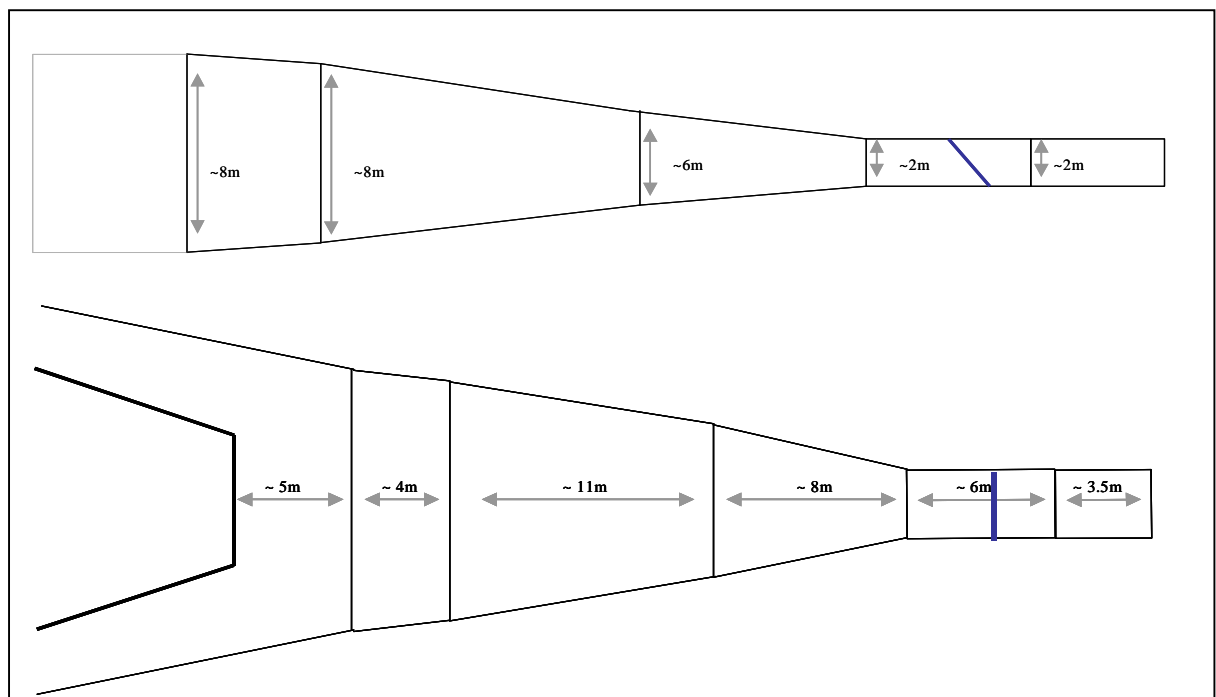
Figure 3: side and top view of net dimensions and excluder grid position in a standard trawl net used in the Pilbara trawl fisher.

It is difficult to understand why a bycatch mitigation strategy in a fishery is working without understanding how such a strategy affects the behaviour of animals interacting with that fishery or the behaviour of the fishing gear. The first aim of this chapter was to use data collected by underwater video cameras and onboard observers

to investigate by which mechanism(s) the deployment of excluder devices has reduced bottlenose dolphin bycatch. In particular, to what extent animals being caught but expelled through the escape hole prior to or during haul back may effect perceived reductions in bycatch rates. The observation by Stephenson and Chidlow (2002) that animals videoed inside trawl nets were orientated towards the vessel means that bottlenose dolphins are unlikely to be orientated in a way that would allow them to detect the bottom opening escape hole easily. Therefore, the second aim of this chapter was to assess the orientation of bottlenose dolphins inside trawl nets relative to the escape hole and excluder grid. The third aim was to investigate the presence of dolphins outside or inside trawl nets and to assess behaviour of bottlenose dolphin inside actively fishing trawl nets through analysis of available underwater video footage.

6.3   Methods

6.3.1   Data

All data were collected by the Department of Fisheries (DOF) Western Australia and were made available for analysis for this chapter.

6.3.1.1   Observer data

Information collected by onboard observers between June 2004 and March 2008, recorded in the DOF Pilbara Trawl Database were utilized to extract data relating to individual fishing operations (depth, time, location, haul duration), type of exclusion grid if deployed, location and orientation of video camera housing and observer comments.

6.3.1.2   Video footage

Underwater video cameras were deployed in the trawl nets as part of a FRDC and then DBIF funded project to assess the performance of excluder grids in the fishery. Video footage was recorded on a Sony HC15E DCR placed in either an aluminium or PVC underwater camera housing which was then attached directly to the net mesh with cable ties. For further details on storage and editing of collected video footage, see Stephenson et al. (2005).

When video footage was taken in a net **with an excluder grid deployed**, the camera housing was positioned in one of the following three locations in the net (Figure 4):

Position A: 2m - 5m downstream of the grid facing upstream

Position B: 2m - 3m upstream of the grid facing downstream

Position C: 2.5m – 4m upstream of the grid facing upstream

When **no excluder grid was deployed**, observers placed the camera facing upstream in a number of positions between 1.5m inside the codend up to 4m in front of the codend lengthener in order to record the behaviour of dolphins inside the net.
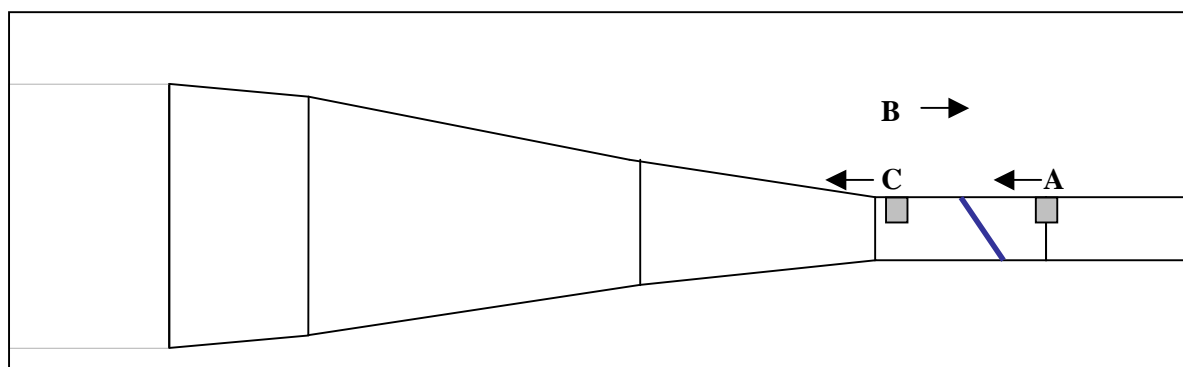
Figure 4: Schematic showing the side view of a standard trawl net in the Pilbara Trawl Fishery. The blue bar indicates the location of the excluder grid and the grey boxes the locations of the cameras. Letters A, B and C and associated arrows indicate the orientation of the camera in relation to the exclusion grid.

All videotapes collected during this time, which still contained original data, were reviewed and categorised according to the location of the camera in the net, the quality of the images obtained and the presence or absence of dolphins in the footage. For tows where video recordings were collected but tapes were reused in the field or subsequently, the Department of Fisheries Pilbara Observer database was used to obtain available missing information.

6.3.2 Data analysis

6.3.2.1 Assessment of bycatch reduction

Data recorded in the PFTIMF observer database were used to analyse bycatch rates in trawl tows in the absence or presence of an excluder grid. These data, supplemented

by additional data obtained from underwater video footage, were used to assess the mechanism(s) behind the observed reduction in bottlenose dolphin bycatch rates in the fishery using a Bayesian approach, where the probability distribution of underlying bycatch rates in the presence or absence of an exclusion grid were compared. All video footage collected from tows where dolphins interacted directly with the exclusion grid and / or dolphin bycatch occurred were reviewed in detail to assess the mechanism by which exclusion grids are reducing dolphin bycatch in the fishery, and to assess the probability of survival of dolphins observed exiting through escape hole in front of the exclusion grid.

6.3.2.2 Metrics used to assess the extent and nature of bottlenose dolphin interactions with trawl nets.

Table 2 summaries data used to assess the extent and nature of bottlenose dolphin interaction with the PFTIMF.

Analysis	Data source
Assessment of bottlenose dolphin interactions with the exclusion grid.	DOF observer database and available tapes collected from locations A, B & C
Interaction rates between dolphins and trawls. Interactions are defined as the presence or absence of dolphins recorded inside or outside the trawl net.	DOF observer database and available tapes collected from locations A, B & C
Encounter rates, duration and individual behaviour of dolphins videoed inside trawl nets.	Highest rated video footage collected by cameras facing upstream (C)
Time of arrival and exit of dolphins from nets.	Video footage collected from cameras facing upstream (C)
Orientation of dolphins inside trawl nets relative to the excluder grid and bottom opening escape hole	Video footage collected from cameras facing upstream (C)

Table 2: Breakdown of data used for different analyses.

Methods used for each of these analyses are described below

Each tape was reviewed in full and scored according to the water visibility inside the net, the proportion of the net circumference visible on screen and the camera location and angle. Only high quality tapes were used to investigate the following, according to the methods described above:

- Interaction rates between dolphins and nets: including dolphins visible outside the net.
- Encounter rate of dolphins inside nets.
- The time at which dolphins were first and last sighted within a fishing operation.

Individual dolphin behaviour inside the net.

6.3.2.3   Encounter rate of dolphins inside trawl nets.

The first and last time a dolphin was visible inside the trawl net during an individual fishing operation (tow) was recorded from video footage collected by cameras facing upstream in the net (Position C). Videotapes collected in the codend of the trawl (Position A) were excluded, as these did not provide an adequate field of view to determine when a dolphin had first reached the throat of the net (the section where the net starts to taper approximately 4-5m upstream of the grid).

The encounter rates of dolphins inside trawl nets was estimated using video footage collected by cameras facing upstream of the grid (Position C), where the visibility on screen was a minimum of 3m forward of the camera and at least two thirds of the circumference of the net could be seen. Minimum encounter rates for dolphins inside nets were calculated from these videos as the total time one of more dolphins were present on screen divided by the total length of the tow recorded on video.

The total length of the tow, which was recorded on video, was taken from the time the net was first seen to be fishing properly until either the tape finished recording or the net had reached the surface during haul back.

The following data were collected for all periods that a dolphin was visible on screen, hereafter termed an *observation*: the time of entry and exit from view, the orientation of the dolphin when first observed and the closest distance it came to the camera during an observation. As there is no way to post-calibrate the videos to provide exact measurements, distances were estimated in relation to dolphin body length and categorised accordingly.

When more than 1 dolphin was present on screen, the time at which the first dolphin entered and last dolphin exited the screen was recorded. This sampling methodology meant some information on individual behaviour in the net was not recorded, as the time of exit was only recorded for the last dolphin visible in the recording, but this was not necessary for obtaining an overview of dolphin encounter rates within trawls.

For each observation the proportion of the net circumference visible and quality of the tape were recorded along with the minimum and maximum number of dolphins visible during the observation. The following data relating to the fishing operation were also recorded directly from the video footage; the time the net was first in the water, the time it started fishing properly, the time haul back commenced and the time the net was closed and at or near the surface of the water.

The initiation of haul back was determined from the video by an observed increase in tow speed prior to haul back, which is an operational characteristic used to flush fish into the codend, or a change in vessel noise or an increase in ambient light as the depth of the net is decreased. All or a combination of these factors may act as cues to the dolphin(s) to exit the net.

6.3.2.4   Behaviour of individual dolphins inside trawl nets.

Behavioural data were recorded from all videos where the activity of an individual dolphin could be observed clearly for the entire duration that it was visible on screen. This means that behavioural data were not recorded for those videos where the presence of an animal could be seen, but low light levels or low visibility meant its

behaviour could not be determined. Behaviour was grouped into one of three broad categories.

Foraging –an individual was observed moving its head from side to side and presumed to be scanning for fish, pursuing fish and / or capturing fish.

Socialising – an individual was observed chasing, touching or displaying synchronous behaviour with another dolphin inside or outside the net.

Unknown – the behavioural state of the focal individual could not be determined.

Only videos where 3 or fewer individual dolphins were present in the net at one time were used, because as the number of dolphins in the net increased it became more difficult to follow individuals. Data on the time of entry and exit, orientation and behaviour whilst in view were also collected from the video footage.

6.3.2.5   Orientation of dolphins inside the net.

Video footage collected by cameras facing upstream in the net (Position C) was used to assess the orientation of individual dolphins as they appeared on screen. Individual orientation was recorded as one of the five categories summarised in Table 3. Additional information on visibility in the net was also recorded.

Orientation Category	Description
Unknown	Recorded when it was not possible to record a dolphin's orientation when it first came into visible range as it was in front of another dolphin.
Tail towards vessel to head towards vessel	The dolphin was first visible swimming head first towards the codend but then turned to drift back so its head was orientated towards the vessel
Sideways across belly	The dolphin was first seen swimming from one side of the net to another but did not back down into the net.
Sideways to head towards vessel	The dolphin was first seen swimming across the net and then turned to drift backwards with its head orientated towards the vessel.
Head towards vessel	The dolphin first appeared backing down the head orientated towards the vessel.

Table 3: Categories of orientation of dolphins when they first appeared on screen inside the net.

6.3.2.6 Assessment of underwater video footage

Camera position, quality of recording and presence or absence of dolphins was determined for each camera deployment by assessing the DOF database, observer notes and available video footage. These data were then used to investigate the interaction rates (dolphin presence or absence) between dolphins and nets, including the presence of dolphins outside the trawl for tows where no dolphins were observed entering the net.

The encounter rate and behaviour of bottlenose dolphins inside actively fishing trawl nets were evaluated from videotapes recorded by cameras facing upstream in the net (position C), and were collected between May 2005 and June 2006. Video footage recorded from cameras facing downstream in the net were not used for this assessment, as in general, only a small portion of the animal (i.e flukes) was visible on screen and was therefore unsuitable for assessing individual behaviour.

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6.4 Results

6.4.1 Overview of video footage collected.

Video cameras were deployed on 575 tows (457 tows with an excluder grid, 118 tows without) in the Pilbara Finfish Trawl fishery between the 1st of January 2005 and the 24th of July 2006, with an additional 5 tows recorded in 2007. Video footage of adequate quality to assess excluder grid performance and/or the presence of protected species in the net was obtained from 65 % (n= 373) of these deployments. A number of factors resulted in unsuccessful camera deployments, including low light and/or visibility in the net, incorrect camera angle or technical problems with the camera.

Video camera deployments were grouped in one of four treatments: facing downstream toward the exclusion grid, facing upstream behind the exclusion grid, facing upstream forward of the exclusion grid, and facing upstream with no exclusion grid in the net. Over 60% of the useable video data was collected from cameras facing downstream towards the exclusion grid.

Each videotape represents a single tow by an individual vessel.

6.4.2 Assessment of the mechanism(s) behind reduced bycatch rates in tows with excluder grids deployed

A total of 4,377 trawl tows were observed, by independent onboard observers, in the Pilbara Interim Finfish Trawl Fishery between June 2004 and March 2008, during which 38 bottlenose dolphin bycatch events were recorded, resulting in the bycatch of 42 individual animals. Excluder grids were deployed in nets for 72% of these observed tows. The use of excluder grids resulted in a significant reduction in the bycatch of dolphins (χ^2=0.018) in this fishery. Table 4 provides a summary of the results of bycatch rates collected on tows recorded by independent onboard observers.

Treatment	No of tows	No of hours towed	No of dolphins caught	Bycatch rate by tows	Bycatch rate by hour
No grid	1243	3062	19	0.015	0.006
Grid	3134	8146	23	0.007	0.003

Table 4: Summary of results of bycatch rates from observed tows.

Table 5 provides a summary of the number of videotapes, by camera position that recorded dolphins present inside the net.

Camera position in the net	Total tapes	No dolphins present on tape	Dolphins present inside the net
Facing upstream	82	28	54
Facing grid*	238	210	28
Behind grid and facing upstream	12	5	7
Unknown**	43	42	1
Total	375	285	90

*Presence ascertained through visual sighting of fluke/dorsal and or sighting of bubbles associated with vocalisation.

** Camera position is unknown as this was not recorded in observer notes and tapes have since been reused.

Table 5: Summary of the presence or absence of dolphins recorded on underwater video.

Dolphins were recorded inside trawl nets in 90 of 375 good quality videos, giving a minimum interaction rate inside trawls of 0.24. In screening the available videos it became clear that camera position was likely to affect the likelihood of observing dolphins if they were to enter the trawl net. Videos recorded by cameras facing towards the vessel, either upstream of the grid, or in the same location without a grid, provided footage of a greater area of the net than those placed downstream of the grid or in the codend. Using only those tapes recorded by cameras facing up stream, the minimum interaction rate of dolphins inside trawl nets in the fishery is 0.66. However, this interaction rate is still likely to be an underestimate as it was not possible to observe dolphins if they were closer to the mouth of the net. This interaction rate also does not take into account whether a grid was deployed in the net.

To determine if the presence of a grid reduced the likelihood of dolphins entering the net, interaction rates were calculated from all videos facing upstream forward of the grid, or recorded by cameras in the same position when no grid was present. Table 6 provides a summary of the data.

	No of tapes (tows)	No of tapes dolphins inside net	% of videoed tows with dolphins inside
Grid	20	12	74%
No Grid	35	26	60%

Table 6: Percentage of videos with dolphins present for each treatment.

There was no significant difference in the number of tapes with dolphins present between the two treatments ($\chi^2 = 0.6393$, p-value > 0.1). Therefore, the presence of the grid alone is not responsible for the observed reduction in bycatch.

Video footage recorded by cameras facing the grid were then analysed to see if interactions between dolphins and the excluder grid (i.e. that dolphins were exiting through the escape hole) explained the observed reduction in bycatch rate. Dolphins were recorded present inside the net in 12% of videos recorded by cameras facing downstream to the grid (n=238). In only 0.03% of these videos was the entire body of a dolphin visible (n=7). Presence of dolphins for the remaining tapes was determined either by part of the dolphin, such as the tail or dorsal fin being visible on screen, or by the presence of bubbles streaming back in the net. Figure 5 summarises the proportion of tapes with dolphins present recorded by cameras facing downstream to the grid.
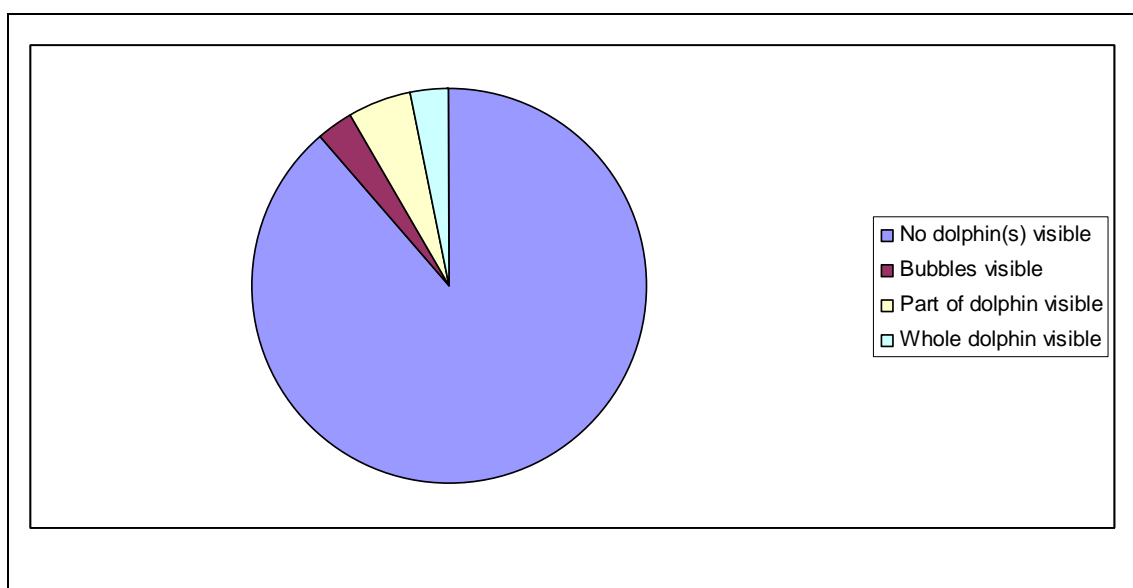
Figure 5: Proportion of videos recorded by cameras facing downstream towards the grid in which the presence of a dolphin was recorded.

Of the 7 dolphins that were recorded on video reaching the grid one is presumed to have swum out of the mouth of the net, as it was not in the net when it was hauled. Four dolphins were observed to exit through the escape hole in front of the grid, and two became caught in the grid. Of the two dolphins caught in the grid only one was landed, with the second falling out of the grid prior to the net being brought onboard.

An additional three videos recorded tows where dolphin bycatch occurred and a grid was deployed. Of these, a camera placed behind the grid and facing upstream recorded one tow and two were from cameras facing downstream to the grid. However, the latter two videos were re-recorded over in the field and could not be analysed. If all nine interactions with the grid are considered, then dolphins exited the escape hole four times, were caught at the grid and landed four times and were caught at the grid but not landed on one occasion. Therefore, the escape rate of four in nine dolphins, exiting from the grid reflects the 50% reduction in bycatch observed in tows with grids deployed. However, if any dolphins die in the net and then are washed out of the grid before the net is towed, bycatch rates for nets with grids will be negatively biased.

In order to test how this bias might affect calculated bycatch rates in tows with grids, all tows where a dolphin was caught in the net but was expelled during haul back were compared to those where an animal was caught in the net and landed. To do this a Bayesian approach, using uniform priors, was used to estimate the underlying probability of bycatch in tows without a grid. 19 dolphins were recorded bycaught in 1,243 observed tows where an exclusion grid was not deployed. These data were used to calculate a posterior distribution of the underlying probability of bycatch in the fishery (Fig 6a). An equivalent distribution was then calculated for all tows with grids deployed where dolphin bycatch was observed, and the animal was landed on deck; i.e. it did not fall out of the net during haul back (Fig 6b).

Figure 6a & 6b: Probability of dolphin bycatch in tows with or without an excluder grid.
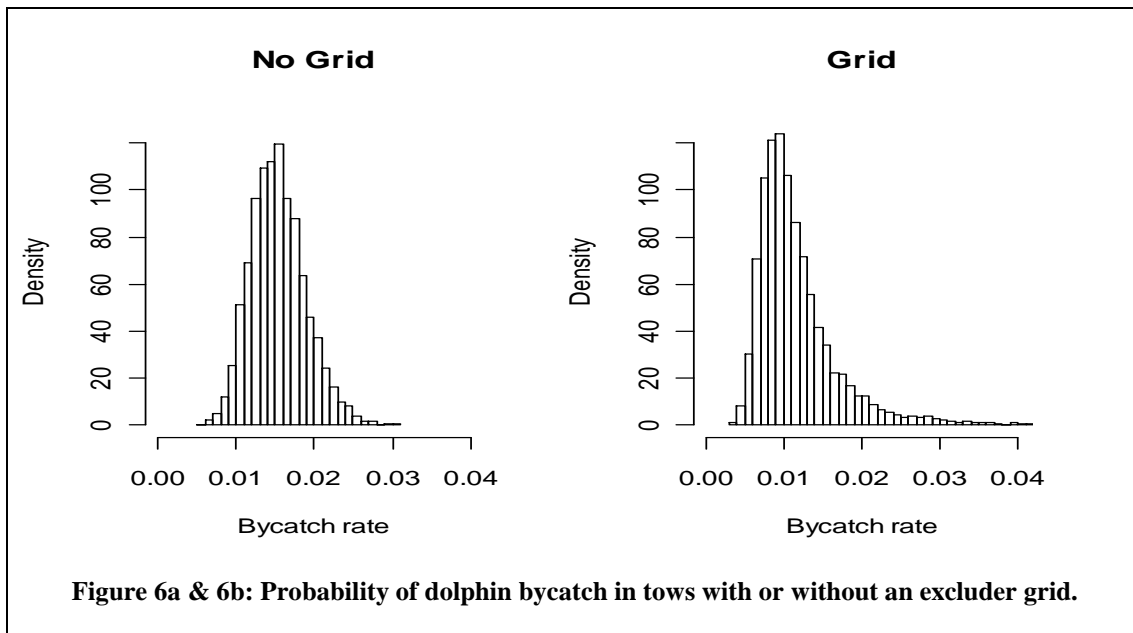
However, data from videos collected in tows with grids deployed showed that one of the five dolphins that died in the net was expelled through the escape hole before haul back. These data were used to calculate the probability of an animal dying in a tow with a grid but not being observed in the net when hauled (Fig 7). Drawing from the posterior distribution for bycatch with a grid when animals were landed, these data were used to calculate a further posterior distribution for the probability of an animal being bycaught but not being landed (Fig 8).

Figure 7: Probability of bycatch occurring but not being landed.

Figure 8: Ratio of bycatch with grid to bycatch with no grid.

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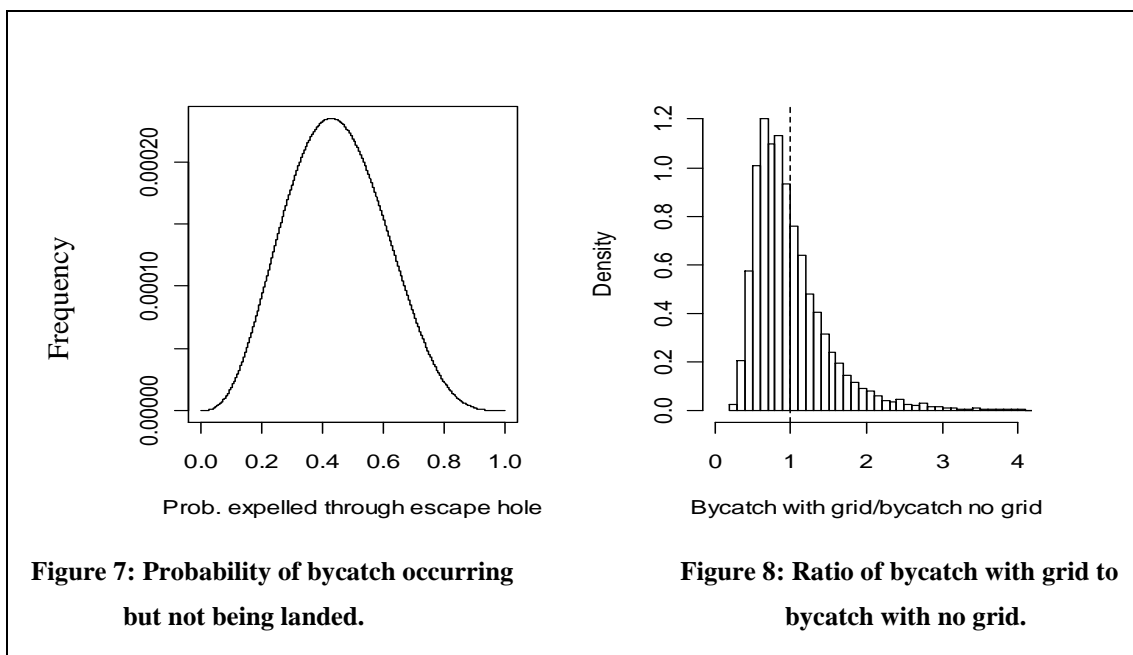
Comparing this distribution to the underlying distribution of the probability of bycatch in a net without a grid gives a probability of 0.8731. Therefore, there is an 87% probability that the 50% reduction in bycatch in nets with a grid is a true reduction and not an artefact of animals being caught in the net but being expelled from the grid before being observed. However, in two of the videos where dolphins escaped through the grid the probability of survival of these animals is difficult to determine. In the first video the animal took over 3 minutes to exit through the grid, while in the second the footage is too dark to determine the state of the animal when it exits the grid. Taking a precautionary approach that three out of seven bycatches the probability of a dolphin dying in a tow but not being landed was used to recalculate the probability of bycatch in a grid to tows without a grid. Figure 9 shows the recalculated ratio of bycatch in tows with the two treatments.

Figure 9: Ratio of bycatch in tows with a grid to tows with no grid, with a landed rate of 4 in every 7 bycaught dolphins.
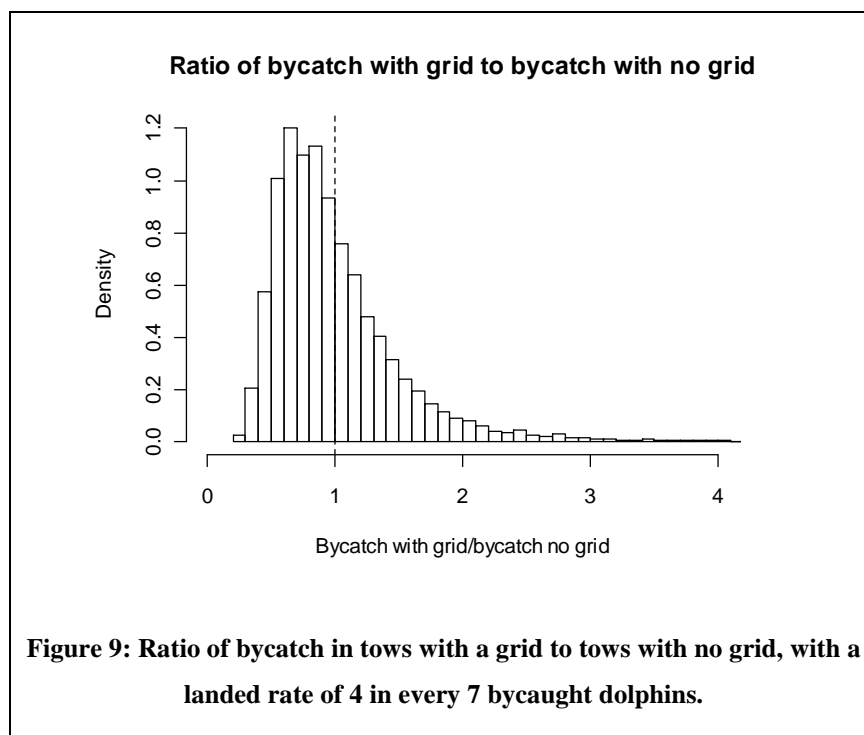
Using this new rate of animals being caught but not landed the probability of bycatch in a net without a grid is 0.6202. This means that we can be 62% sure that the 50% reduction in bycatch in nets without a grid is a true reduction and not an artefact of animals being caught in the net but being expelled from the grid before being observed.

6.4.3 Bycatch rates by vessel

A high variability in bycatch rates of non-target species by individual vessels has been reported (Dietrich and Fitzgerald 2010). Therefore, bycatch rates of the 6 vessels observed in the fishery were investigated. Table 7 provides a summary of bycatch rates by vessel.

Vessel	No Grid			With Grid		
	No. of tows	No. of dolphins	Bycatch Rate	No. of tows	No. of dolphins	Bycatch rate
A	53	1	0.019	651	0	0.000
B	17	0	0.000	212	2	0.009
C	468	8	0.017	765	13	0.017
D	589	7	0.012	847	3	0.004
E	116	3	0.026	522	4	0.008
F	No data	No data	No data	137	1	0.007
Totals	**1243**	**19**	**0.015**	**3134**	**23**	**0.007**

Table 7: Bycatch rates by vessel for tows with and without a grid.

While the bycatch rates observed for vessel B were higher when a grid was deployed, the number of tows observed on this vessel when no grid was deployed was very low (n=17). The bycatch rates of all other vessels, except vessel C, were reduced by the deployment of grids.No tows were observed in vessel F prior to the deployment of an exlucder grid, so no background bycatch rates can be estimated for this vessel. While overall bycatch rates in this fishery had been reduced by approximately 50%, if vessel C and F are removed from the analysis then the use of excluder grids has reduced bottlenose dolphin bycatch rates in this fishery by over 70%. (Table 8).

Vessel	No Grid			With Grid		
	No. of tows	No. of dolphins	Bycatch Rate	No. of tows	No. of dolphins	Bycatch rate
A,B,D,E	775	11	0.014	2232	9	0.004

Table 8: Bycatch rates for vessels combined (excluding vessel C and F).

In order to determine whether there was an equal probability of dolphins entering each vessel's nets, 82 videos recorded by cameras facing upstream were assessed. Tapes that were recorded behind the grid were not included in the analysis because of the variability in the quality of the visibility of the net from videos recorded in this position. Table 9 shows the proportion of videos where dolphins were observed present inside the net per vessel and the total number of tows videoed.

Vessel	Proportion of videoed tows with dolphins inside	Total number of videoed tows
A	0.77	22
B	0.71	7
C	0.76	17
D	0.55	22
E	0.86	14

Table 9: Proportion of videoed tows per vessel from cameras facing upstream which recorded dolphins inside the net.

The number of tows with dolphins varied significantly among vessels (χ^2= 11.4179, p-value <0.05). However, this difference is driven by the low number of dolphins recorded in tows by vessel D. If this is removed from the analysis then there is no difference among the other four vessels (χ^2= 3.4482, p>0.1). Therefore, the higher bycatch rate by vessel C is unlikely a result of dolphins being more frequently inside nets of this vessel.

Video data and observer notes were examined to determine what may have caused the higher bycatch rates for this vessel when grids were deployed. Table 10 provides a summary for each vessel, where this information was available, on where in the net dolphins were caught. No information was available on the position of three of the 23 bycaught animals.

Vessel	Caught in Grid/ fell through the escape holeat tow back	Forward of the grid	Position unknown
B	2	0	0
C	8	2	0
D	1	1	1
E	2	1	1

Table 10: Position of dolphin caught in tows when a grid was deployed.

These data show that 68% of dolphins that were bycaught in tows with grids were either caught in the grid itself or fell out through the grid when the net was towed on deck. Vessel C had the highest proportion of dolphins caught in the grid or falling through the grid during haul back. Of the dolphins that were caught forward of the grid, one was tangled in the headline, one was tangled in a line inside the trawl net

and one bycatch occurred after the net snagged, collapsed and was subsequently towed.

6.4.4 Bottlenose dolphin behaviour inside trawl nets.

37 videos recorded by cameras facing upstream towards the vessel and collected between May 2005 and June 2006 were then analysed in more detail to assess the general behaviour of dolphins inside the trawl nets. Figure 10 shows the orientation of dolphins from 346 observations recorded by cameras facing upstream. No dolphin was observed tail towards the vessel by the time it was within 1 dolphin length of the camera. In general dolphins were observed to turn in the "belly" section of the net (the wide area forward of where the net begins to taper), and were already positioned head towards the vessel by the time they reached the section where the net narrows.
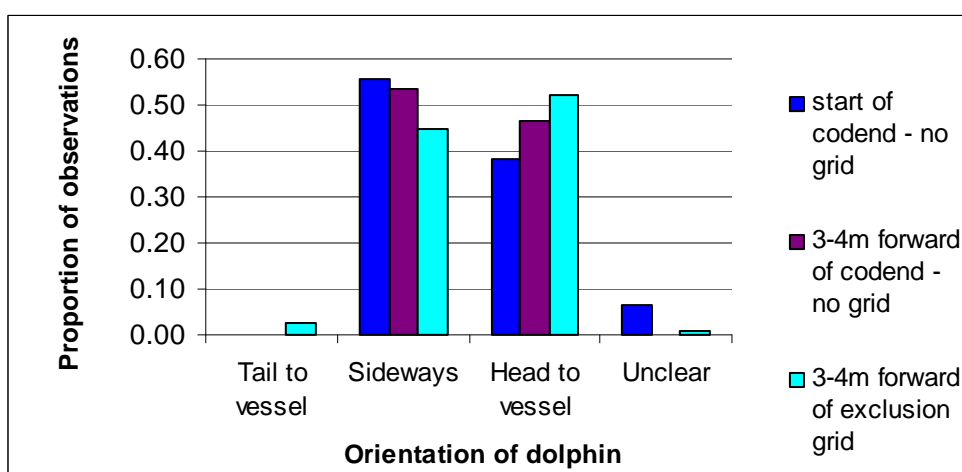
Figure 10: **Orientation of dolphins inside trawl nets.**

Conversely, in two videos recorded by cameras facing the grid, one dolphin was seen to approach head first towards the grid and then exited head first, and one dolphin landed sideways across the grid before exiting head first. In general, however, dolphins do not appear to be orientated in a way that would allow them to detect the bottom escape opening easily.

Additionally, the furthest distance that dolphins drifted back in the net appeared to be related to the position of the camera in the net, when cameras were upstream of the

grid or no grid was present. These data are presented in Table 12 in relation to whether a grid was or was not present in the net.

Treatment	1 dolphin length in front of camera	In front of camera	Dorsal fin on/under/behind camera
Grid	10	9	24
No Grid	71	69	18

Table 12: Counts of the furthest back dolphins were recorded for tows with and without a grid.

The furthest point that dolphins drifted backwards in the net was significantly different between tows with and without a grid ($\chi^2 = 53.11$, p-value = 0.0005). When there was no grid in the net, the camera was placed either where the grid would have been or further back towards the codend. Dolphins remained further ahead of the camera when there was no grid. However, for some of these tapes the exact location of the camera was difficult to ascertain. When a grid was deployed, the camera was a minimum of 3m forward towards the vessel than when there was no grid. One note by an observer stated that when the camera was just in front of the codend animals were observed to drift back into the codend.

These observations suggest that presence of a grid, or a camera without a grid prevent dolphins from drifting back all the way to the codend. It is likely that the closer dolphins are to the mouth of the net the higher the likelihood that they will be able to exit through the mouth when the net changes shape for example at haul back. Therefore as well as the observed reduction in bycatch which can be assigned to dolphins exiting through the escape hole, it is possible that the presence of a physical barrier, whether camera or grid, increases the chances of a dolphin being able to exit through the mouth of the trawl as the net is collapsing. Observer notes report that in most of the instances when dolphins were caught in nets without grids they were not physically entangled in meshes of the net, which possibly suggests that animals being caught as the net collapsed rather than being entangled in the net prior to haul back. To investigate if the presence of a camera without a grid in the net may also reduce bycatch by preventing dolphins drifting back, bycatch rates for the four treatments (grid, no grid, camera, no camera) were calculated (Table 13).

Treatment	Tows	No of dolphins	Bycatch rate
Camera no Grid	72	0	0.000
Grid - No camera	3400	17	0.005
Grid + Camera	316	4	0.013
No Grid - No Camera	1147	19	0.017

Table 13: Bycatch rate of bottlenose dolphins in four different treatments.

As already reported, the highest bycatch rates occurred in trawls without grids or cameras. Although, the lowest bycatch rates occurred in tows with cameras but no grids deployed, the samples size (n=72) is too small to draw any conclusions. The higher bycatch rates in tows with grids and cameras suggest that bycatches may be underestimated in tows with grids and no cameras.

37 videotapes, recorded by cameras facing upstream in the net (position C), were used to investigate the temporal occurrence of bottlenose dolphins inside trawl nets within a fishing operation. The average time between when the net was recorded as fishing properly and the first dolphin was observed inside the net was 14 minutes (min=1,max=60, mode=1) (Figure 11).
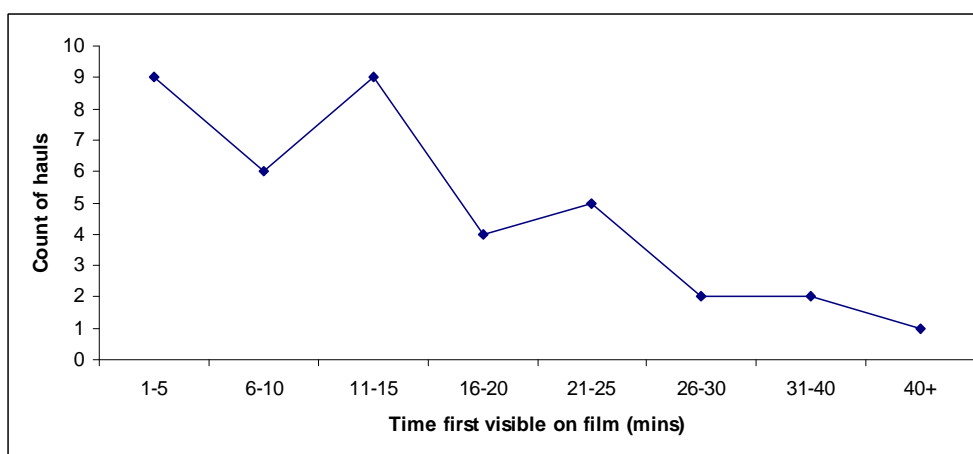
Figure 11: First time of arrival in minutes of a dolphin inside the net.

Pearson rank correlation was used to test the correlation between the time of arrival in the net and the sequence of the tow in the trip; no correlation was found (P=0.212).

Tow back of the net was recorded in 16 of the 32 tapes. The last dolphin was observed exiting the net 2 minutes or less before the net collapsed in 9 of these 16 tapes

(min=1, max=11.5, mode=2)(Fig 12). This indicates that at least some individuals are able to exit the net just before tow back is complete without getting caught.
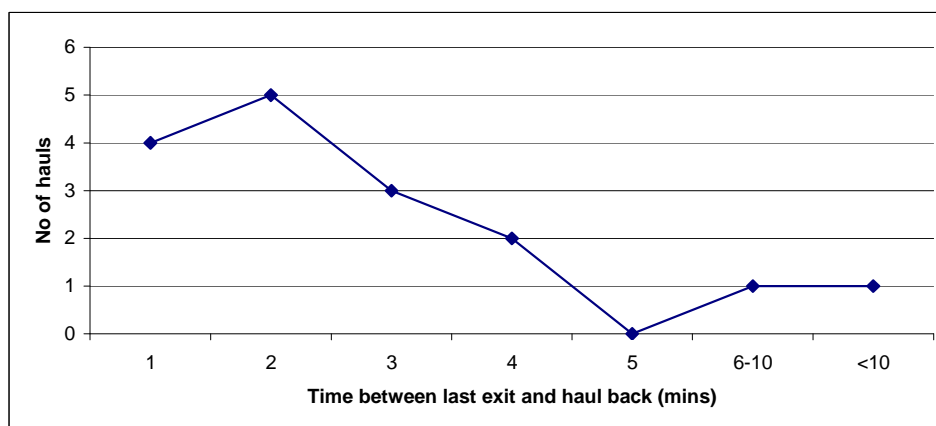
Figure 12: Last time of exit of a dolphin from the net relative to haul back of the net.

6.4.5   Encounter rates of bottlenose dolphins observed inside trawl nets

There was a positive correlation between the average proportion of times dolphins are visible in the net and the maximum group size recorded (Pearson's Rank Correlation, P=0.96). Figure 13 shows a plot of the time since the previous observation of a dolphin against the proportion of observations made for each group size. This plot and the positive correlation reflect the observation that when more than one animal is in the net there is a constant rotation of position in the net.
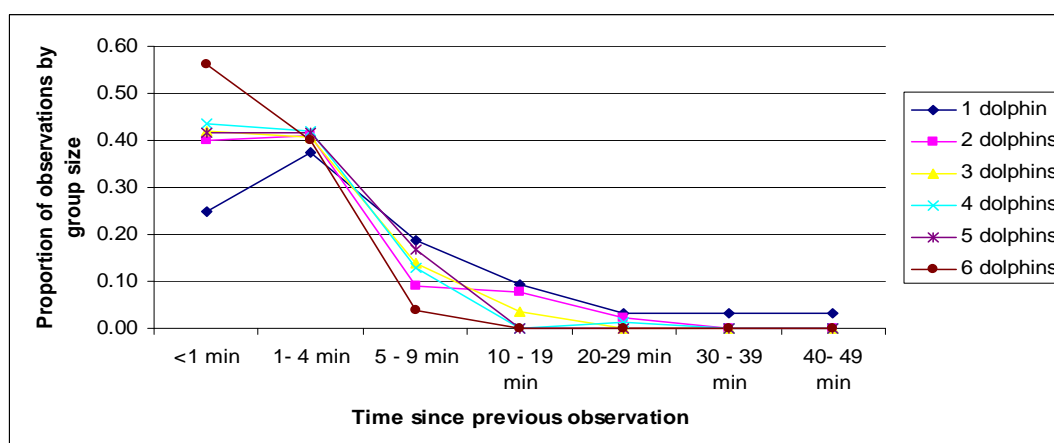
Figure 13: Time recorded since previous observation.

6.4.6 Individual foraging behaviour

Low visibility because of high sediment level in the trawl or low light levels meant that although individuals could be classified as foraging, direct observations of fish capture were relatively rare and only observed in 18% of all foraging observations. Of these observations, 34 definite fish captures (fish seen in mouth) and 28 possible fish captures (scales seen in water, change in body posture associated with fish capture) were observed. Figure 14 shows the number of fish caught per sampling period per individual animal. Each sampling period was the total time an individual dolphin was inside the net and visible in the video.
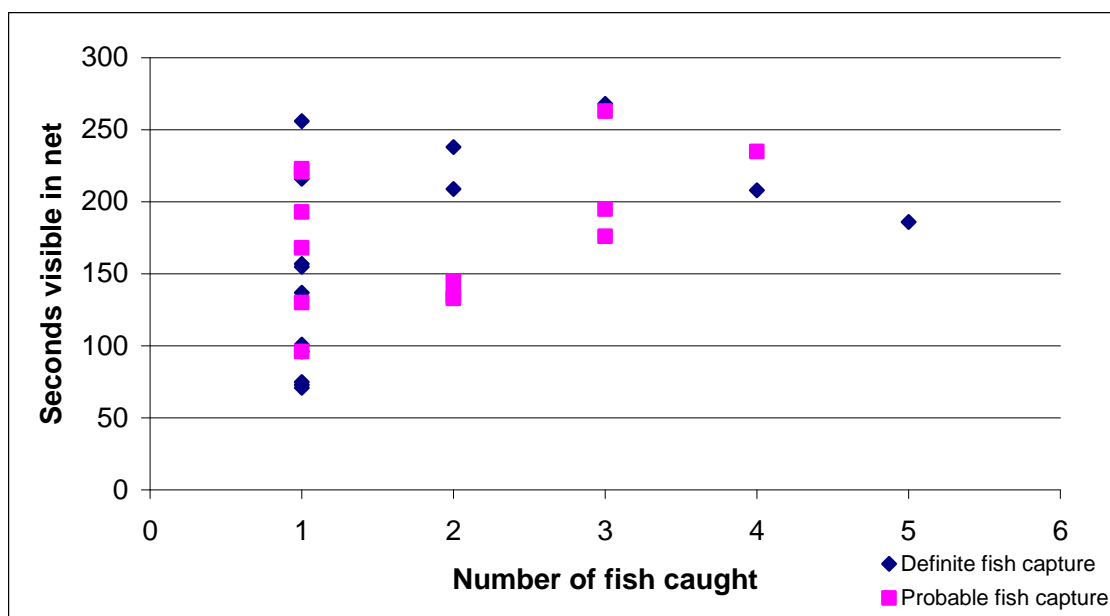
Figure 14: Fish capture rate by individual by individual dolphin.

Data on CPUE were only available at the level of the trip and not for the individual tows in which animals were observed foraging. Figure 15 shows the definite and probable capture rates plotted against this averaged CPUE; however, these data do not account for inter-tow variability in catch rates.

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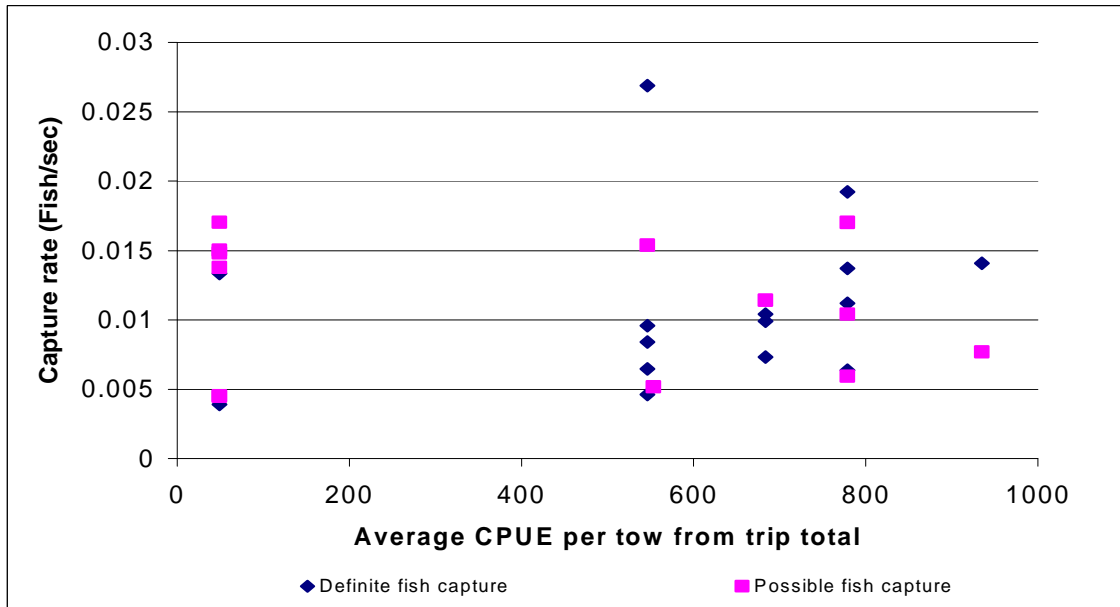
Figure 15: Fish capture rate by individual dolphin against CPUE (averaged per tow from trip data).

Figure 16 shows a plot of fish capture against maximum group size for that tow. Fish capture and group size were negatively correlated (Pearson's Rank correlation P= -0.02067). However, these data do not account for the actual numbers of dolphins that were in the net when a fish capture or probable fish capture event took place.
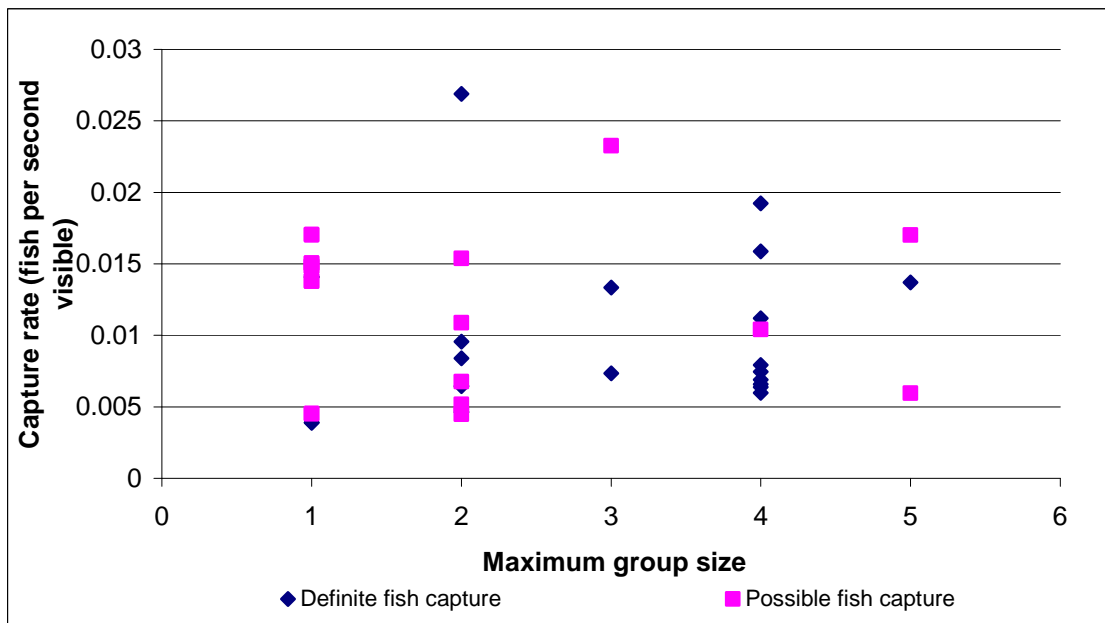
Figure 16: Number of fish captured by individual against maximum group size for that tow.

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While data collected by underwater videos analysed in this trial were generally unsuitable to record the number of dolphins present outside a net, it was possible to establish that 34% of tapes recorded from tows facing upstream dolphins were visible on the outside only, indicating that dolphins were often present outside nets but do not always enter them.

6.5   Discussion:

6.5.1   Mechanism(s) behind the reduction of bottlenose dolphin bycatch

The bycatch of bottlenose dolphins has been significantly reduced by the deployment of excluder grids in the Pilbara Trawl Fishery. Results from an analysis of the probability of dolphins being bycaught but expelled through the escape hole prior to or during haul back, showed that there is an 87% probability that the observed reduction in bycatch rates is real. While video footage recorded four animals exiting through the escape hole, it was difficult to determine the condition of two of these animals. Taking a precautionary approach and assuming that these animals would not have survived, the probability of the observed bycatch reduction rate of 50% was recalculated as 62%.

A reduction in bottlenose dolphin bycatch rates was not observed for one vessel (Vessel C), where bycatch rates were equal in tows with or without a grid. However, there was no significant difference in the proportion of tows with dolphins present inside nets. However, there was insufficient power to determine if there was any difference in the maximum group size observed between this and four other vessels for which video footage was available. Information was available on the position where entanglement occurred for 18 of the 23 dolphins bycaught in the presence of a grid. Of these 18 animals 68% were found to have been caught at the grid, or fell through the escape hole during haul back, with 80% of dolphins caught by vessel C falling into this category. A number of studies have noted that whilst different vessels within a fleet may appear to behave exactly the same, particular vessels may continuously record the highest bycatch rates (Du Fresne et al. 2007, Dietrich and Fitzgerald 2010). In the case of the Pilbara Finfish Trawl Fishery, it is possible that consistently high bycatch rates recorded by vessel C may have been a result of the way in which the exclusion grid was deployed in the net of this vessel. Observer notes showed that this vessel had repeated problems with the angle of the grid whilst it was fishing, and a number of alterations were tried with respect to the flap used to prevent fish loss through the exclusion grid. In addition, data were not available on the type of grid used (solid or flexible) for all analysed hauls. Therefore, the influence of grid or escape hole type on bycatch rates could not be tested. The existence of vessels with

above average bycatch rates can increase the variance and uncertainty in bycatch estimates if sampling design results in higher or lower observer coverage of such vessels. In addition, such variation between vessels can compound the results of mitigation trials and it is important that the effects of the deployment of excluder devices (or gear modifications) on the fishing behaviour are properly assessed (Chapter 3).

A review of the seven videos that recorded direct interactions between bottlenose dolphins and the exclusion grid showed that in two events a dolphin was able to exit the net alive through the escape hole. However, the remaining five recorded events indicated that the current configuration of the grid in the net did not provide an easy pathway for dolphins to exit, especially as they were generally orientated head first towards the vessel. In addition the downward positioning of the escape opening will result in an underestimation of bycatch rate in this fishery, because bycaught animals are likely to fall out of the escape opening during haul back.

Because the incidence of dolphins recorded exiting the net through the exclusion grid is low, it is possible that the mechanism by which exclusion grids are reducing bycatch is by acting as a barrier and preventing dolphins from drifting too far back in the net. In June 2008 exclusion grids were moved forward toward the vessels to the start of the net extension, in order to provide a shorter distance between the grid and the opening of the net (Allen & Loneragan 2010). However, during this trial three dolphins were recorded interacting with the two grids deployed in these positions, two of which were reported to fall through the escape hole when already dead, while one was not excluded. A fourth dolphin was recorded bycaught in the head rope of the trawl net. Due to the design of the grid and the escape hole, the orientation of the animal when it encounters the grid affects the likelihood of escape. Allen & Loneragan (2010) note that at least one of the dolphins they observed was hampered from exiting the net through a bottom-opening hatch. Trials of a top opening hatch were conducted in March 2009, however, only 30 hauls with this modification were observed (Allen & Loneragan 2010).

In the UK pelagic bass pair trawl fishery some bycaught common dolphins were observed orientated head first towards the codend whilst other were orientated

towards the vessel. Underwater video footage in this fishery showed that dolphins that were orientated towards the codend when they reached the grid were able to exit through the upward opening escape hole. However, in both these fisheries, on occasion dolphins were observed to arrive at the grid already in an exhausted or catatonic state and were therefore unable to find the escape hole.

6.5.2   Interactions between bottlenose dolphins and the Pilbara Trawl Fishery

From the underwater video footage collected during the excluder grid assessment trials (2005 –2007) it is clear that the interaction rate between bottlenose dolphins and actively fishing trawl nets is high. Video footage recorded from cameras facing upstream in the net showed dolphins entered into the trawl nets in a minimum of 66% of tows and could be present up to 64% of the total duration of the net being in the water. However, due to the position in the net of the cameras which recorded these videos, these results are likely to be underestimates of the encounter rates of dolphins inside these tows. The behaviour of bottlenose dolphins in and around nets in these trawls has since been analysed further using additional video footage, recorded in 36 trawls since June 2008 after exclusion grids were moved forward in the net towards the vessel (Jaiteh 2009). These data were collected by video cameras orientated towards the vessel at a distance of 3.6m upstream of the new grid position. These data provided much clearer video footage of dolphins both inside and outside of the trawl net, and dolphins were observed outside the net during 94% of tows and inside the net during 81% of tows.

6.5.3   Behaviour of bottlenose dolphins around and inside trawl nets

While the quality of video footage varied as a result of ambient light and amount of sediment in the water, a number of videos provided insights into the behaviour of bottlenose dolphins inside and in the vicinity of nets in this fishery. These data show that individuals are very aware of the nets and are not averse to touching them from both inside and outside the net. On a number of occasions dolphins outside the net were observed to remove enmeshed fish, and in some footage could be seen actively

foraging underneath the belly of the net as the benthos was stirred up by the ground rope.

A common behaviour of dolphins outside the net throughout the duration of the tow and during haul back was "bouncing" on the net as it was being towed. A related behaviour observed in conjunction to bouncing was "resting" on the net, where a dolphin outside the net would lie on the top panel of the trawl net as it was being towed for 10 – 30 seconds. When this behaviour was observed it was common if a dolphin was concurrently inside the net, that animal would orientate itself belly upwards and press against the upper surface of the net and hence the dolphin outside.

Most observations of dolphins inside nets showed the animals were actively engaged in foraging and were utilizing one of two general strategies. The first involved chasing fish in the belly of the net and the other more commonly caught on video because of where cameras were positioned showed animals positioning themselves head first to the vessel at the start of the exclusion grid (or codend) extension where they would then use the surrounding net as a barrier to capture fish against. Jaiteh (2009) also reported a high proportion of foraging behaviour exhibited by dolphins inside the trawl net (56%).

It is interesting that although a group of six animals was observed in one tow, in general observations were made of one animal in the net at a time, and this did not seem to be related to the number of animals that were directly observed outside the net or were inferred to be there through "bouncing" on the net. While the data analysed for this study were not available to confirm this observation, Jaiteh (2009) found that dolphins were observed outside the net in 5 hauls but did not enter the net, while a total of 29 individually identified dolphins were recorded inside nets in the remaining 36 hauls. She reported that while some individual animals were generally present in the trawl alone, five identifiable pairs of dolphins were seen inside the net, each pair in two separate hauls. The largest group of identifiable dolphins she recorded foraging inside a trawl net was seven. From the data analysed for this study, it was clear from identifiable dolphins that individuals would enter and exit the net on a number of occasions within the duration of the haul. Jaiteh (2009) also found

individually identifiable dolphins entered and exited the net on multiple occasions during a single haul.

From repeated sightings of individually identifiable dolphins within a single tow and between subsequent tows, it is likely that foraging inside trawls is a specialization by some animals and results of analysis of videos collected further towards the vessel appear to support this (Jaiteh 2009). In particular, she found that, with the exception of one identifiable animal, all other individuals were re-sighted in tows within a specific area and hypothesized that dolphins may associate with trawlers when they are present in the home range of an animal, rather than following trawlers throughout the fishing area. If resident populations and trawl fisheries consistently overlap it is likely that more animals will learn to exploit nets. To compound this, if repeated trawling of the same ground results in depleted fish stocks, animals may increasingly be attracted to trawls to forage. In Moreton Bay, Australia, two distinct communities of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) occur. These communities share an overlapping distribution, but are socially segregated and exhibit different foraging behaviour, with one community regularly observed foraging around commercial prawn trawlers (Chilvers and Corkeron 2001, Chilvers et al. 2003). Investigation into the behaviour and spatial distribution of these dolphins found that they preferentially foraged with trawlers even when fishing effort was reduced. Nearly 80% of their diurnal behaviour during the study was spent foraging in association with trawlers. However, it is interesting to note that relative to other studies of inshore bottlenose dolphins, "non trawler dolphins" were also recorded to spend a high proportion of time foraging, and the authors suggest this may be an indication that prey availability in the area may be reduced (Chilvers et al. 2003). Bottlenose dolphins have also been reported interacting with bottom trawl boats along the Mediterranean continental shelf of Israel (Scheinin 2010), and that encounter rates were significantly higher when survey effort was targeted around bottom trawlers. Jefferson (Jefferson 2000) also reports that only certain individual Indo-Pacific humpback dolphins in Hong Kong waters interact with trawls whilst others were never observed to do so.

Analysis of stomach contents of bycaught Australian fur seals in the winter trawl fishery off Tasmania, Australia, found that animals were feeding on blue grenadier; a

species that can only be accessed by seals when it is brought into their diving range by the trawl net (Tilzey et al. 2006). Tilzey et al. concluded that there was little evidence from this analysis to support the idea that seals were undertaking any substantive foraging away from the trawl nets. In addition, seals that had been caught from trawlers and satellite tagged showed a noticeable change in the distribution of their foraging effort once the fishing season ended. These studies indicate that some marine mammals actively associate with trawl nets in order to forage inside or around these nets. However, such associations may be specializations for individuals or groups or animals rather than the whole population in areas where marine mammals and trawl fisheries regularly overlap.

Bottlenose dolphins were observed actively foraging inside trawl nets in this and Jaiteh's study (2009). Highest observed fish catches occurred when dolphins were positioned in the narrow part of the net and used the net as a barrier to catch fish against. It is therefore likely that these dolphins choose to back down in the net as far as possible to areas were they can still manoeuvre but they can use the shape of the net to aid in foraging. In the UK pelagic pair trawl fishery for bass some footage was obtained of dolphins feeding in front of an exclusion grid on two occasions (>1hr and >40 mins respectively) indicating that one or more individuals repeatedly swam to that part of the net to forage (Northridge and Mackay 2005). In contrast, some dolphins observed inside nets in the Pilbara Trawl Fishery never came within 2 dolphin lengths of the camera and could be observed swimming in the belly of the net only.

Whilst no bycatch was recorded in tows where a camera but no exclusion grid was deployed, the sample size was too low to infer that the deployment of a camera alone might have reduced bycatch rates. Jaiteh (2009) also reported that none of the 30 dolphins she observed backed down past the video camera when it was positioned further forward in the net. However, bycatches were still recorded in this fishery when exclusion grids were placed further forward in the net (Allen & Loneragan 2010), but the authors noted that at least one of the dolphins would have had a greater chance of exiting the net had a top opening escape hole been present.

Analysis of video data collected in this fishery show that dolphins are clearly able to enter and exit the net with ease as it is fishing. Therefore, the likelihood of bycatch may be increased as a result of the net fishing incorrectly. One dolphin was observed bycaught in a net that came fast on the sea floor and subsequently collapsed. However, Jaiteh (2009) observed one dolphin successfully entering and exiting a net that never fully opened throughout the duration of the tow, and concluded that bycatches are more likely to occur when young or inexperienced animals enter the net. It is clear that while exclusion grids have reduced the overall bycatch rates of bottlenose dolphins in this fishery, improvements in grid design, position and orientation of escape holes are likely to further reduce these incidental captures.

6.5.4   General Discussion

The evaluation and field-testing of bycatch mitigation devices is often hampered by a lack of understanding into the behaviour of marine mammals around fishing gear (Read 2005). The results of this chapter clearly show that it is difficult to understand why a bycatch mitigation strategy is working without understanding how animals are behaving towards it or how this strategy may effect the behaviour of the fishing gear. If animals are motivated to feed inside nets it is important to understand their orientation in the net and which areas of the net they may preferentially forage in. In addition, the design, development and testing of gear modifications requires constant communication between fishermen and gear technologists. Although exclusion grids became mandatory in the Pilbara Trawl Fishery in 2006, it is clear from observer notes and through discussions with fishermen in March 2008, that problems remained with the deployment of grids and positioning and size of escape holes. However, details relating to modifications were not systematically recorded in the observer data, and therefore no assessment of the effect of different modifications on bycatch rates could be made. It is clear that ongoing real time assessments of how modifications affect the behaviour of fishing gear is required, as is an open sharing of knowledge among members of the fishing community.

It is also important to understand which operational characteristics of a tow may reduce or increase the likelihood of interaction and /or entanglement. However, the

fact that trawl shots in such trials cannot, by their nature, be exact replicates must be kept in mind, even if factors such as depth, location or tow speed are kept consistent (Tilzey et al. 2006). The speed at which the net is hauled may influence the bycatch rate and skippers in the Pilbara conducted some investigation into this but the results were equivocal (K. Head, *pers. comm.*). In Tasmania the results of a model of the bycatch rates of two trawlers suggested that duration of haul back may be an important factor in explaining their differences in bycatch rates (Tilzey et al. 2006). Although it is clear from the Pilbara data that dolphins can become bycaught during normal towing of the net, it is possible that they are more susceptible to entanglement during hauling of the net. However, it is also clear that dolphins can become entangled in other parts of trawl nets during normal towing, such as the "lazy line" (codend retrieval line) of shrimp trawls. An evaluation of interactions between bottlenose dolphins and such lines by the NMFS Southeast Fisheries Science Centre utilizing DIDSON sonar found that dolphins were encountered during every tow and each time they were recorded interacting with the lazy line by rubbing against it as they exited the net area (Anon. 2009). Even though interaction rates were high, no entanglement was observed. In the Pilbara fishery a skipper experimented with a bungee cord stretched across the width of the trawl extension piece to which was attached a number of cable ties. In one video, a dolphin was seen to swim quickly forward after its fluke touched the device but in a subsequent tow an animal was found entangled in the bungee so the experiment was abandoned (P. Stephenson, *pers. comm.*). Another factor that may influence bycatch rates is whether nets are towed in a straight line or include turns (Smith and Baird 2005); in many fisheries trawl nets are partially hauled while vessels turn. Smith and Baird (2005) reported that the bycatch rate of female New Zealand sea lions was 2.33 times higher in tows which had a turn, compared to those which were fished in a straight line. However, no such relationship between bycatch rates of male New Zealand sea lions and tow characteristics were observed.

The observed higher bycatch rates in tows when both a camera and exclusion grid were deployed compared to tows when just an exclusion grid was deployed suggest that bycatch rates may be underestimated if they can not be confirmed through video monitoring, because it is impossible to record the number of animals that successfully exit the net via an escape hole. In addition, in some studies (Tilzey et al. 2006, Lyle

and Willcox 2008) seals have been observed to enter the net via the escape hole to forage on fish in front of the exclusion grid. There is the added uncertainty of whether those animals that do successfully escape through an escape hole will subsequently survive. As previously stated there are few other studies where underwater observations of marine mammals interacting with trawl nets have been recorded, and these are particularly limited for cetaceans. The results of this chapter show that a greater understanding of the behaviour of bottlenose dolphins inside trawl nets, through the collection of underwater video, can be used to improve the design and application of bycatch mitigation strategies.

6.6 References

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7. Chapter 7: General discussion

7.1 Synthesis

The incidental capture, or bycatch, of large marine vertebrates species in fisheries operations threatens the conservation status of a number of these taxa. Interactions between non-target species and fisheries are likely to increase as a growing human population results in more intensive harvesting of the oceans. The threat of bycatch is particularly grave for small populations that are unable to sustain even small numbers of removals of individuals from the population (D'agrosa et al. 2000). Bycatch has been linked to the decline of a number of large marine vertebrates including sea turtles (Peckham et al. 2007, Gilman et al. 2010), marine mammals (Read et al. 2006, Read 2008), sharks and rays (Baum et al. 2003) and birds (Zydelis et al. 2009a). Monitoring and mitigation of large marine vertebrate bycatch has generally focused on commercial or industrialised fleets. However, over 90% of the worlds fishermen are thought to be employed in artisanal fisheries (Peckham et al. 2007, Zydelis et al. 2009b, Moore et al. 2010) and there is growing concern over the possible cumulative impact of such small-scale coastal fisheries may have on non-target species (Read et al. 2006, Read 2008, Soykan et al. 2008, Gilman et al. 2010). Numerous examples of high bycatch rates of large marine vertebrates in such fisheries already exist (D'agrosa et al. 2000, Amir et al. 2002, Alfaro-Shigueto et al. 2007, Peckham et al. 2007), and there is evidence that, for some species, the impacts of artisanal fleets may be higher than those of industrialised fleets (Peckham et al. 2007).

Bycatch reduction may be achieved through a number of management strategies including time/area closures (Murray et al. 2000) or an overall reduction in fishing effort. However, changes in fishing gear and practices are thought to have the highest potential of reducing the bycatch of large marine vertebrates in global fisheries (Werner et al. 2006, Zydelis et al. 2009b).

In order to determine how fishing gears or practices might be modified to reduce the incidental capture of large marine vertebrates, factors affecting bycatch rates of non-target species within a specific fishery must be identified (Haas et al. 2008). Statistical analysis of data collected by independent observers can be used to provide insights

into which gear characteristics may have the potential to be modified to reduce bycatch of non-target species (Palka 2000), to interpret the results of gear trials (Watson et al. 2005), and to assess the continued efficacy of adopted gear modifications or fishing practices on bycatch rates (Watson et al. 2005, Gilman et al. 2007, Gilman et al. 2008).

7.1.1 Identifying potential gear modifications to reduce harbour porpoise bycatch in UK bottom set gillnets.

In Chapter 2, generalized linear models (GLM) were used to identify which factors influence the bycatch rates of cetaceans in UK commercial set net fisheries, using independent onboard observer data collected over a 14-year period. In particular, the aim was to see whether any specific gear characteristics were associated with high or low bycatch rates, and whether there is any potential to modify these characteristics to reduce cetacean bycatch in static net fisheries. Data from gillnets and tangle nets were initially combined for two distinct fishing areas, the North Sea (ICES Area IV) and the South West of Britain (ICES Area VII). For ICES Area IV the only significant predictor of harbour porpoise bycatch was fleet length, which had a negative relationship with bycatch rates. This relationship was driven by high bycatch rates in cod wreck net fisheries, where the majority of observed hauls were in fleets less than 500m in length. For ICES Area VII the only significant predictor of harbour porpoise bycatch was net height, which had a positive relationship with harbour porpoise bycatch rates. This relationship was driven by high bycatch rates in gillnets targeting whitefish, in nets with an average rigged height of 5.2 metres. While the results of these models identified specific fisheries with high bycatch rates, they did not provide information on which characteristics within these, and other, fisheries might be appropriate to investigate for their potential to mitigate bycatch. For this reason separate models were constructed to investigate bycatch rates of harbour porpoise in gillnets and tangle nets for ICES Area IV and VII.

Fleet length was found to have a significant negative relationship with harbour porpoise bycatch rates in both gillnets and tangle nets in ICES Area IV. Bycatches in gillnets in this area were observed in wreck nets fishing for Cod (*Gadus morhua*) with highest bycatch rates recorded in nets less than 500m. High bycatch rates in wreck

nets may be due to higher densities of harbour porpoise prey species around wrecks or in the general area of wrecks, or the close proximity that wreck nets are set together in an area. Likewise, highest bycatch rates in tangle nets were observed in fleets of less than 150m in length. The retention of this variable in the best tangle net model was likely driven by two separate experiments that utilised relatively short fleet lengths. However, this result is confounded by higher average soak times and higher average deployment depths of these nets relative to other tangle net experiments conducted in the same area. As harbour porpoise bycatch was modelled as a rate, using an offset of km net hours per haul, the retention of this explanatory variable indicates that the assumption of a proportional relationship between harbour porpoise bycatch and fishing effort does not always hold true. Vinther (1999) when using km net hours as a measure of effort calculated highest bycatch rates in Danish cod fisheries, however, when bycatch rates were modelled using fleet length as a measure of effort the highest rates were in the Danish turbot fishery, as net soak durations in this fishery are twice those of the cod fisheries he investigated. In contrast, the best model for predicting harbour porpoise bycatch in ICES Area VII, did not retain fleet length as an explanatory variable. Given that bycatch rates were calculated per km net hour of fishing effort, this result reflected the fact that the highest bycatch rates in this area were observed in short wreck nets for cod and longer gillnets targeting hake and other whitefish. While fleet length was also retained for the best model of tangle nets in Area VII the relationship was not significant. In addition, this model was found to be unstable when resampled.

Mesh size was retained as a covariate in the best models for predicting harbour porpoise bycatch in gillnets in both the North Sea (ICES Area IV) and South West of the UK (ICES Area VII). The relationship between mesh size and bycatch was positive, with higher bycatch rates occurring in larger mesh sizes. This positive relationship with bycatch rates has previously been shown for harbour porpoise (Palka 2000, Orphanides 2009), bottlenose dolphins (Palka and Rossman 2001) and loggerhead turtles (Murray 2009). Mesh size was not retained in the best model predicting harbour porpoise bycatch in tangle nets ICES Area IV, but only two mesh sizes were observed in tangle nets used in experimental trials. In ICES Area VII the retention of the explanatory variable floats present in the best model may have been a

proxy for mesh size, however, in this area no harbour porpoise bycatch was recorded in nets with the largest mesh sizes.

The results of two separate experimental trials in the UK suggested that twine diameter may play a role in influencing harbour porpoise bycatch rates (Northridge et al. 2003), and that the reduction of harbour porpoise bycatch in thin twined nets may in part be due to the reduced breaking strain of the meshes, as well as the smaller mesh size used. No bycatches were observed in sole gillnets in the South West, although 24% of observer effort was in this fishery. This fishery is characterized by an average mesh size of 121mm made of thin twine and shorter net heights. Whether the lack of bycatch is due to the thinner twine material of these nets or their lower net profile cannot be ascertained from the data. Vinther (1999) also observed no bycatches in the Danish sole fishery and concluded that this could be due to the thinner twine used in this fishery as well short soak duration and mesh size used.

There was a significant positive relationship between harbour porpoises bycatch rates and rigged net height in gillnets in ICES Area VII. Highest bycatch rates were recorded in wreck nets and gillnets targeting hake with rigged heights of 4.1 to 6 metres. While the mesh size of sole nets is similar to those used in gillnet targeting cod and hake the rigged height of these nets is much lower, averaging 1.6m. In contrast, rigged net height was not retained as a significant predictor of harbour porpoise bycatch in gillnets in ICES Area IV. However, 87% of observed gillnet hauls in this area were in wreck nets and longer cod nets, both of which had an average rigged height of 3.6m.

Results of an analysis of factors influencing the bycatch rate of common dolphins were less clear, suggesting that the bycatch rates of this species in bottom set nets may be driven by a temporal and spatial overlap of animals and fishing gear, rather than specific characteristics of that gear. For example, common dolphin bycatch was recorded in two very different gear types (a cod wreck net with a rigged height of 7.8m, and a large mesh tangle net with a rigged height of 1.5m), deployed by the same boat in the same area.

The results of Chapter 2 indicates two possible routes of gear modification that should be tested to determine whether they might prove a useful mitigation strategy for (at least) harbour porpoise bycatch. These are a reduction in the rigged height of bottom set nets, and the use of thinner twine material. A reduction in fishing profile of gillnets in the USA by the addition of tie downs has been associated with a reduction in harbour porpoise bycatch (Palka 2000), and reducing the breaking strain of gillnet meshes has been suggested previously as a gear modification that should be tested (Northridge et al. 2003, Read et al. 2003). However, as of yet, no paired experimental trials have been conducted to assess directly the effect of reduced rigged height or twine diameter on small cetacean bycatch rates. Tie downs are required in a number of USA bottom sink gillnet fisheries under the Harbour Porpoise Take Reduction Plan (HPTRP), and the length of tie downs, and required spacing are specified. However, a review of compliance between June 2008 and June 2009 found 30% of observed fisheries in the Mid-Atlantic, which were required to use tie downs, were non-compliant (Orphanides 2010). The HPTRP has recently been amended to include a change in the spacing of tie downs on large mesh gillnet fisheries in the Mid-Atlantic (Orphanides 2010). Although the use of tie downs reduce the fishing profile of a gillnet, as a result they also increase the "bagginess" of the net, and while their use has been associated with a reduction in harbour porpoise bycatch, they have been associated with increased bycatch rates of other protected species. The elimination of tie downs and the use of lower profile nets has been shown to reduce the bycatch of sea turtles whilst maintaining acceptable levels of target catch (Price and Van Salisbury 2007), and low profile nets have been shown to reduce bycatch rates of Atlantic cod (He 2006). A recent study found that while the elimination of tie downs reduced the bycatch of Atlantic sturgeon (ICES 2011), an increase in marine mammal bycatch was also observed in these nets. Therefore, while lowering the profile of fishing nets by using tie downs has been shown to reduce the bycatch rates of harbour porpoises, in order to concomitantly reduce bycatch rates of other species, it would be useful to test whether such a reduction could be achieved by reducing the overall rigged profile of the net whilst still maintaining target catch levels.

7.1.2 Investigating the effects of gear modifications on the underwater fishing behaviour of static gillnets.

There still remains a lack of understanding about the actual fishing heights of static gillnets, although it has been shown that the relationship between fishing height and rigged height will be affected by a number of variables including the amount of flotation used, fleet length, mesh size and twine diameter, as well as the speed of the water current (Stewart & Ferro 1985). Nonetheless, these factors have rarely been considered in paired bycatch mitigation trials, and tools for measuring them (both practical and statistical) are not well developed. In Chapter 3 this issue was addressed in the context of the results obtained from previously conducted paired trials (Northridge et al. 2003) to see how information on the relative fishing behaviour of control and modified nets used in such trials could alter conclusions on observed bycatch rates.

Data were collected on the fishing behaviour of five bottom set nets with differing gear characteristics using self-contained depth loggers. This study confirmed that the active fishing height of the float line of bottom set nets was affected by the amount of flotation, mesh size and twine diameter of the gear, as had previously been observed by Stewart & Ferro (1985). The proportion of net area fished relative to the theoretical net area of these five gillnets ranged from 22% to 65%. For three gillnets of equal length, mesh size, twine diameter and rigged height, the proportion of theoretical net area fished was greatly affected by the amounts of flotation and lead line used. The net rigged with double flotation and lead line weight fished 65% of its rigged profile, the standard rigged net fished 48% of its profile, while the net rigged with a float line of polypropylene rope and cigar floats fished only 22% of its rigged height. In addition, a standard gillnet for bass was found to fish on average 45% of its theoretical net area, while a gillnet modified with $BaSO_4$ fished 36% of its theoretical net area. The rigged height of the $BaSO_4$ net was 1.75 times that of the standard skate net (Single net). Therefore, considering only rigged height, to see a 50% reduction in bycatch rates in the $BaSO_4$ net, the number of animals caught in these nets would need to be 88% of that caught in the standard net. However, the mean fishing height of the $BaSO_4$ net was 1.44 times that of the standard skate net, therefore, if all other things were equal, in order to achieve a 50% reduction in bycatch rates there would

need to be 72% the number of porpoises caught in the modified net compared to the standard skate net.

To date, no study has directly assessed the effect of net height on bycatch rates of cetaceans in bottom set gillnet fisheries, although the use of tie downs which lower the fishing profile of gillnets are associated with lower bycatch rates of harbour porpoises (Palka 2000). Therefore, if the probability of cetacean entanglement is reduced in gillnets with lower fishing profiles, then the results of the proportion of net area fished by the five gillnets investigated in Chapter 3 indicate that if the fishing heights of gillnets used in paired trials are not equal, than the results of such trials will be ambiguous. Results of a simulated experiment showed that the number of animals caught in control and experimental nets would be considered significant if both nets were assumed to fish with equal net profiles. But these bycatch rates would no longer be significantly different if the reduction in net profile of the experimental net, and therefore reduction in the probability of bycatch, was accounted for. Therefore, the results of paired trials to assess the potential for gear modifications to reduce bycatch rates of non-target species are difficult to interpret when more than one variable is changed (Read 2007).

During the design stage of paired trials, power analyses are generally conducted to determine the number of hauls required to provide sufficient power to detect a (specified) significant reduction in bycatch rates in modified fishing gear. However, if a modification of gear leads to an indirect reduction in the theoretical fishing area of the net, resulting in a decrease in the probability of bycatch occurring, then the number of hauls required to detect a specified reduction in bycatch, due to the specific gear modification, will be underestimated. Results of a power analysis for a theoretical fishery, conducted in Chapter 3, showed that the number of hauls required to have 80% power to detect a bycatch reduction of 50% increased from 215 hauls of both control and modified nets, to 302 hauls of both treatments, if the gear modification used in the experimental nets inadvertently led to a 30% reduction in the fishing profile of this net type, and a concurrent 30% reduction in the probability of bycatch occurring.

In order to maximize the utility of bycatch reduction trials and allow results to be tested and compared in other locations, then the effects of gear modifications on experimental nets relative to control nets need to be examined in parallel to any bycatch mitigation trial. This is especially important in order to be able to make inferences about how or why non-target species are caught in the fishing gear.

7.1.3 Investigating the relationship between bycatch rates and the behaviour of modified fishing gear in an experimental trial.

In order to directly assess the fishing behaviour of modified gillnets and whether these data could be used to inform conclusions about observed bycatch rates, depth loggers were deployed on one control and two experimental types of net used in a bycatch mitigation trial in Argentina (Chapter 4). This mitigation trial aimed to assess the utility of reflective ($BaSO_4$) and stiffened nets at reducing the bycatch rates of Franciscana dolphins, an endemic species to the region that is impacted by high and probably unsustainable bycatch rates (Secchi et al. 2003). Analysis of depth logger data from three deployments of the control, stiff and $BaSO_4$ nets showed that the $BaSO_4$ net fished with a mean float line height which was significantly lower than the other two nets. Over the three deployments the proportion of the theoretical net area fished by the three nets was 40% - 50% for the $BaSO_4$ net, 60% - 70% for the stiff net and 50% - 70% for the control net. However, this observed reduction of the fishing area of the $BaSO_4$ gillnet did not result in a concurrent reduction in Franciscana bycatch rates in these nets. A statistical analysis on the independent onboard observer data collected during the bycatch mitigation trial found that the only significant predictor of Franciscana bycatch was latitude. In addition, the reduction in fishing profile of the $BaSO_4$ net did not result in a reduction in target catch rates as there was no significant difference in overall catch per unit effort in the three nets.

While the use of stiff or acoustically reflective nets does not seem to hold much promise in reducing the bycatch of Franciscana dolphin in bottom set gillnet fisheries, nets with similar properties have significantly reduced the bycatch rates of harbour porpoises in two other paired experimental trials (Larsen et al. 2007, Trippel et al. 2009). The contrasting results of these studies may be due to the fishing behaviour of nets during these trials or the different morphology of the species involved. In the trial

by Trippel et al. (2009) no additional flotation was added to gillnets modified with $BaSO_4$. The results of Chapter 4 show that these nets fish lower than standard nets, and that the orientation of the nets to the current, and the current speeds they are subjected to, are likely to increase the scale of this reduction. Current speeds in the area where Trippel et al. (2009) conducted their trial can reach up to 4 knots (Cox et al. 2004) and it is possible that the observed reduction in bycatch rates was due to the reduction in net profile of these nets relative to the standard nets used in this trial. In contrast Larsen et al. (2007) added additional flotation to the experimental nets modified with iron oxide (IO) that they used in their trial. As well as a significant reduction in harbour porpoise bycatch in IO nets in this study, there was also a significant reduction in the both the total catch and size length of cod caught in these nets. The reduction of harbour porpoise bycatch in these nets was attributed to the increased stiffness of the IO nets. The lack of reduction in Franciscana bycatch in either the stiff or $BaSO_4$ net may be due to the morphology of these animals. During the experimental trial in Argentina, entangled dolphins frequently had their rostrums, which are slim and elongated, fully wrapped in the net with gillnet meshes between their teeth. In addition, although there is little information on the acoustic behaviour of these animals, it has been hypothesised that they may not echolocate while travelling between feeding areas (Bordino et al. 2002), which would make them more susceptible to entanglement. Although CPODs were deployed during the experimental trial in Argentina they failed to work. As a result there remains no information on the echolocation behaviour of Franciscana in the vicinity of gillnets.

The ability to design gear modifications or assess the effectiveness of their implementation is often confounded by a lack of understanding of the behaviour of the bycaught species in the vicinity of fishing gear (Chapter 4). Approaches to mitigating interactions between non-target species and fisheries may differ if entanglements occur simply because there is an overlap in the distribution of the fishery and non-target species, or if the bycaught species is motivated to interact with the fishery. Although a substantial amount of effort has been spent on the development and testing of marine mammal bycatch mitigation strategies, especially in Europe, Australia, New Zealand and the USA (Werner et al. 2006), the mechanisms by which individual cetaceans become entangled in fishing gear remain poorly understood, and the evaluation and field-testing of bycatch mitigation devices

is often hampered by a lack of understanding of the behaviour of marine mammals around fishing gear (Read 2005). Therefore, the aims of Chapters 5 and 6 were to investigate the behaviour of cetaceans around fishing gear.

7.1.4 Echolocation behaviour of harbour porpoises in the presence and absence of bottom set gillnets.

Studies utilising passive acoustic monitoring (PAM) have shown that harbour porpoises are in the vicinity of bottom set gillnets much more frequently than bycatch occurs (SMRU et al. 2001, Cox and Read 2004). It has been hypothesised that harbour porpoises may be attracted to struggling fish caught in static gillnets (Gaskin 1984), but this hypothesis has not been tested. Therefore, the aims of Chapter 5 were to determine whether echolocation rates of harbour porpoises were higher in the vicinity of gillnets compared to when no net was deployed, to determine if echolocation rates or behaviour were affected by the gear characteristics of the net and whether data collected by PAM could be used to determine if harbour porpoise are foraging in the vicinity of nets.

Harbour porpoise echolocation clicks were recorded each day of the 51 day study in Bridlington Bay, North Yorkshire. There was no significant difference in the echolocation encounter rate, encounter length or encounter intensity recorded by PODs deployed with or without a net. But, the proportion of fast click trains was significantly higher when a net was deployed and there was a significant positive relationship between the proportion of fast click trains in an encounter and the length of an encounter. Harbour porpoises have been shown to use fast click trains during navigation (Verfuß et al. 2005) and during prey capture (DeRuiter et al. 2009), therefore the higher proportion of fast click trains recorded when a net was present could either indicate that harbour porpoises were inspecting the nets or were foraging around the nets. Kindt-Larsen (Kindt-Larsen 2007) analyzed the stomach contents of bycaught harbour porpoise and hake captured during the same haul in a commercial static gillnets fishery in Denmark. She found no significant overlap in prey items in the stomachs of porpoise or hake, although sample sizes were small. Clearly, further

evidence is needed to confirm that harbour porpoise are actively foraging around gillnets.

The risk of entanglement to a foraging individual may be higher under a number of scenarios. The target strength of the prey item it is approaching may mask echoes from the webbing of the net, and therefore the porpoise may not detect the net, resulting in entanglement. Alternatively, the porpoise may not concentrate on the closeness of a net in the final moments of prey persuit. Harbour porpoises have been observed to forage by "bottom grubbing". During this foraging behaviour an animal positions its rostrum close to the seabed, focusing its echolocation clicks downwards (Stenback 2006). It is clear that animals engaged in such behaviour would have a lower likelihood of detecting a bottom set gillnet before entanglement would occur, or during foraging may get closer than intended to a net it has previously detected. Rising click rates in sequences of echolocation trains of harbour porpoises have been attributed to harbour porpoises adapting their bio sonar to account for the two-way travel time of an echolocation click as they approach a landmark (*N. Tregenza pers. comm.*). The presence of such landmark sequences were found when PODs were deployed with or without a net, indicating that harbour porpoises in Bridlington investigated the PODs with their biosonar. It is unclear how much of the data on echolocation behaviour recorded during this or other studies was a result of harbour porpoise acoustically inspecting the PODs. While this and previous studies (SMRU et al. 2001, Cox and Read 2004) have shown that harbour porpoise occur much more frequently around gillnets then they are caught, all these studies have used PODs and it is difficult to deduce whether it is the presence of these PODs, which possibly stimulated harbour porpoise to acoustically inspect the area of the trials in more detail that resulted in the low bycatch rates observed.

It is clear that in order to assess the behaviour of marine mammals around fishing gear it is important that the method does not inadvertently change the behaviour of the animals being studied. Previous trials using video cameras, either deployed from blimps to give an aerial view of behaviour (Read et al. 2003, Hodgson et al. 2007) or deployed underwater with fishing gear (Konigson 2007, Campbell et al. 2008) have proved highly informative regarding the behaviour of marine mammals around fishing gear or their behaviour response to pingers. Given the frequency with which harbour

porpoises were recorded in the study area in Bridlington bay, the use of a suitable underwater camera, in union with passive acoustic data loggers with a lower target strength than PODs, would be useful in future to further determine the behaviour of this species around gillnets.

7.1.5 Investigating the behaviour of bottlenose dolphins interacting with a bottom trawl fishery.

Footage collected by underwater video cameras deployed inside actively fishing bottom trawl nets proved useful for assessing the behaviour of bottlenose dolphins (*Tursiops spp*.) interacting with these nets (Chapter 6). Data collected by these cameras, in conjunction with data collected by independent onboard observers, were analysed to determine by which mechanism(s) the deployment of excluder devices has reduced bottlenose dolphin bycatch in the Pilbara bottom trawl fishery, Australia. In addition, these data were used to assess the behaviour of bottlenose dolphins inside actively fishing trawl nets. The bottlenose dolphin is the cetacean species most often documented feeding in association with trawl nets (Fertl and Leatherwood 1997, Broadhurst 1998, Chilvers and Corkeron 2001, Chilvers et al. 2003, Pace et al. 2003, Svane 2005, Fortuna 2006, Gonzalvo et al. 2008, Anon. 2009). However, in many instances where associations between animals and nets are common, bycatch rates remain low (Chilvers and Corkeron 2001).

The efficacy of excluder devices at minimizing marine mammal bycatch has been tested in a number of fisheries (Gibson and Isaksen 1998, Northridge et al. 2003, Browne et al. 2005, Northridge et al. 2005, Tilzey et al. 2006, Lyle and Willcox 2008). These devices became mandatory in the Piblara bottom trawl fishery, Australia, on the 1st of March 2006 after they had been shown to be successful in reducing the bycatch rate of bottlenose dolphins (Stephenson et al. 2006). However, it was unclear whether this observed reduction was overestimated, as observers had noted that a number of bycaught dolphins had been expelled from the escape hole during haul back and some had been videoed dead in the net, but were no longer in the net when it was hauled. The results from the analysis of underwater footage collected inside actively fishing trawl nets in this fishery combined with independent

onboard observer data indicated that, even taking a precautionary approach, there was a 62% probability that the 50% bycatch reduction rate observed was correct.

Interaction between bottlenose dolphins and this fishery are high, with bottlenose dolphins observed to enter the trawl nets in 66% of tows where video footage had been recorded. A study that analysed more recent data collected by video cameras placed further towards the mouth of trawl nets in the same fishery, reported that dolphins were inside trawls in 81% of monitored tows (Jaiteh 2009). Most observations of dolphins inside nets showed that the animals were actively engaged in foraging and were utilizing one of two general strategies. The first involved chasing fish in the belly of the net and the other, more commonly caught on video because of where cameras were positioned, showed animals positioning themselves head first to the vessel at the start of the exclusion or codend extension where they would then use the surrounding net as a barrier against which to capture fish. Jaiteh (2009) also reported a high proportion of foraging behaviour exhibited by dolphins inside trawl nets in this fishery (56%). From repeated sightings of individually identifiable dolphins within a single tow and between subsequent tows, it is likely that foraging inside trawls is a specialization by some animals and results of analysis of videos collected further towards the vessel appear to support this (Jaiteh 2009).

A number of studies have noted that whilst different vessels within a fleet may appear to behave exactly the same, particular vessels may continuously record the highest bycatch rates (Du Fresne et al. 2007, Dietrich and Fitzgerald 2010). Results of Chapter 6 indicated that the use of an excluder grid did not reduce bottlenose dolphin bycatch rates for one vessel in the fishery. Analysis of underwater footage collected inside actively fishing trawl nets showed no significant difference in the proportion of tows with dolphins present inside nets, or the maximum group size observed between this and four other vessels for which video footage was available. It is possible that consistently high bycatch rates recorded by this vessel may have been a result of the way in which the exclusion grid was deployed in the net of this vessel. Observer notes showed that this vessel had repeated problems with the angle of the grid whilst it was fishing and a number of alterations were tried with respect to the flap used to prevent fish loss through the escape hole in front of the exclusion grid. These problems led to frustrations for fishermen and crew about the use of excluder devices.

Such frustrations were also experienced when turtle excluder devices (TEDs) were introduced in the USA Gulf of Mexico shrimp trawl fishery (Campbell and Cornwell 2008). The implementation of efficient bycatch mitigation strategies requires continued monitoring and enforcement to ensure compliance by fishermen (Cox et al. 2007). There also needs to be a system that allows the continued development and fine-tuning of gear modifications or technical measures to reduce bycatch, which should be undertaken collaboratively between fishermen, gear technologists and scientists. In the Pilbara fishery, only one skipper had previous experience fishing with an excluder grid. Continued open dialogue and timely problem solving needs to occur if technical measures such as excluder devices result in fish loss or increased difficulty in handling gear, otherwise opposition by fishermen to the use of such devices may result in their misuse or even non-use. While fishermen in the Pilbara did actively experiment both with the angle of the excluder device and the material used to cover the fish escape hole there was no systematic record of these changes, and therefore their effect on bycatch rates recorded by onboard observers could not be determined. This study shows the importance of the continued development, testing and monitoring of bycatch mitigation technologies.

7.2 Conclusions

The results of this thesis show the importance of understanding how modifications aimed at reducing the bycatch rates of non-target species affect the fishing behaviour of experimental gear types (Chapter 3 and Chapter 4) and the need to have a greater understanding of the behaviour of non-target species interacting with gear (Chapter 5 and Chapter 6). Technological advances have lead to the development of a number of loggers that proved useful in assessing the underwater behaviour of fishing gear and how this may relate to observed bycatch rates (Gamblin et al. 2007, McFee et al. 2007, Shimizu et al. 2007, Brillant and Trippel 2009). The use of video cameras and passive acoustic monitoring can provide information on the behavioural responses of free ranging marine mammals interacting with fishing gear (SMRU et al. 2001, Read et al. 2003, Konigson 2007, Campbell et al. 2008), while passive acoustic monitoring can be used to assess the occurrence rates of echolocating cetaceans around fishing

gear (SMRU et al. 2001, Cox and Read 2004) or behavioural responses to bycatch mitigation technologies such as pingers (Koschinski 1997, Cox et al. 2001, Hodgson et al. 2007). However, it is important that any tool used to assess the behaviour of non-target species does not inadvertently affect their behaviour (Chapter 5).

A better understanding of how modifications affect the fishing behaviour of gear as well as how the species in question reacts to such changes is needed to understand by which mechanism bycatch is reduced. This requires a suite of different methodologies to be used and Campbell et al. (2008) provide an elegant case study. Mitigation of seal lion bycatch in an Australian fishery for rock lobster was achieved by using satellite telemetry data to identify areas of high sea lion and fishery overlap, surveys and analysis of voluntary log books were used to assess the scale and nature of the interactions, a sea lion excluder device (SLED) was developed in conjunction with local fishermen, and the behaviour of sea lions interacting with the SLED was assessed using underwater observations. This need to understand by which mechanism bycatch is reduced is especially important if a given bycatch reduction method appears to work for one species in a specific gear type but does not work for another. This was the case for the experimental trial conducted in Chapter 4, where no reduction of Franciscana bycatch was observed in either stiffened or $BaSO_4$ nets (Chapter 4) although a significant reduction of harbour porpoise bycatch had previously been reported in these modified nets (Trippel et al. 1999, Larsen et al. 2007). These results confirm that solutions to bycatch, through the modification of fishing gear, are likely to be fishery specific and will depend on which species is interacting with the fishery. A review by Read (2007) showed that circle hooks have the potential to reduce the bycatch rates of marine turtles in some but not all longline fisheries. In particular, they are unlikely to be effective for species which actively depredate bait from hooks such as loggerhead turtles (Read 2007), and the effects of circle hooks on target catch varied in the four studies reviewed. Therefore, mitigation trials with sufficient power to prove the efficacy of gear modifications should be conducted in fisheries before the uptake of a specific gear modification is advocated, even though it may have reduced bycatch in another fishery.

Data collected from independent onboard observer programmes can be utilised to identify which gear or fishing practices result in high bycatch rates (Palka 2000), and

an analysis of such data collected in UK static net fisheries suggest that net height and twine thickness are two characteristics of gillnets which should be modified to test their efficacy at reducing bycatch (Chapter 2). However, the results of this chapter also showed that in order for explanatory models to have enough power to identify specific gear characteristics, or fishing practices, that were related to higher bycatch rates, a sufficient number of bycatch events in individual fisheries are required. This is particularly the case where over-parameterisation of models can occur if the number of observed events per number of parameters is low (Peduzzi et al. 1996). The analysis of UK observer data found that the ability of models to find such relationships is confounded by relative rarity of bycatch events and the distribution of observer effort in fisheries where no bycatches are recorded along with small coverage in some metiers.

While independent observer programmes are essential to provide robust and unbiased estimates of bycatch rates of non-target species (Read 2010), such programmes are costly and may be impractical in small scale fisheries (Gilman et al. 2010). In addition, the continued monitoring and enforcement of fisheries requires substantial financial investment. Mandatory 100% observer coverage in the Hawaiian long line fishery for swordfish and mandatory 25% observer coverage in the Hawaiian long line fishery for tuna is estimated to cost US$3 million per annum (Zydelis et al. 2009b).

There is a growing body of literature which suggests that the global cumulative bycatch of large marine vertebrates in small-scale artisanal fisheries is a serious problem, yet there is little to no information of bycatch rates in these fisheries and there is an urgent need to address this knowledge gap (Peckham et al. 2007, Read 2008). Detailed catch and effort data are usually not available for such fisheries and conservation resources are limited, therefore, quick and cost effective techniques are needed to identify which fisheries may have high bycatch rates. A recent assessment of large marine vertebrate bycatch in a number of artisanal fisheries around the world was conducted using questionnaires (Moore et al. 2010). Results showed the ubiquitous nature of marine mammal and sea turtle bycatch in artisanal fisheries. Although the authors noted that while such interview surveys could not be used to estimate bycatch rates, they do provide a useful low cost technique to identify those fisheries on which bycatch monitoring efforts should focus.

For both industrialised and artisanal fisheries, uptake of any bycatch mitigation strategy will rely on the economic cost being low, or the cost of non-compliance being high. Bycatch strategies with high economic costs may work if the cost is low relative to the overall profitability of the fishery. However, artisanal fisheries are primarily conducted in developing nations (Peckham et al. 2007), and can have substantial social and economic importance. It is therefore essential to consider the costs associated with any bycatch reduction strategy and how it will affect the economic and social welfare of fishermen (Gilman et al. 2010). While modified gillnets were not found to reduce the bycatch of Franciscan dolphins in an experimental trial conducted in an artisanal fishery in Argentina (Chapter 4), the use of pingers has previously been successful in significantly reducing bycatch rates of this species (Bordino et al. 2002). However, pingers are expensive both to implement and enforce and are unlikely to be a feasible solution for small-scale artisanal fisheries in this region. Therefore, in the case of the Franciscana the use of other fishing gears such as hook and line may provide a more cost effective method of reducing bycatch.

Numerous studies and reviews cite the importance of the direct participation of fishermen in the development of bycatch mitigation strategies as well as the need for a collaborative approach between scientists, managers, conservationists and the fishing industry (Lewison et al. 2004, Cox et al. 2007, Peckham et al. 2007, Campbell and Cornwell 2008, Campbell et al. 2008, Soykan et al. 2008, Gilman et al. 2010). A number of successful bycatch mitigation technologies have been developed by, or in conjunction with, fishermen and there is no doubt that fishermen possess a vast well of knowledge that can greatly inform scientists and gear technologists. A notable example was the development of the Medina panel, a panel of smaller mesh within a purse seine, which increased the escape rate of dolphins during back down procedure in the eastern tropical Pacific tuna fishery (Hall 1998). In addition, it is important to understand the broader social, economic and political dimensions of specific fisheries to ensure that bycatch mitigation technologies with the highest probability of uptake by fishermen are identified (Campbell and Cornwell 2008). Peckham et al. (2007) demonstrated that by including fishermen and local communities in an open dialogue about the importance of conservation of non-target species, communities can view these species as a valuable resource and make cooperative decisions about their

conservation and management. The authors give the example of fishermen in an artisanal fishery in Mexico declaring an area of fishing ground, which had been identified as a core area for loggerhead turtles using satellite telemetry data, as a "Fishers' Turtle Reserve".

In order to identify quickly potential modifications to gear or fishing operations which could be utilised to reduce bycatch rates, and to have the power to sufficiently test these, there needs to be a relatively high rate of incidental capture of non-target species. However, for highly threatened species with small populations, it is not possible to conduct such experiments and the only conservation solution that remains is to remove all risk of entanglement. However, such action is highly expensive and politically unpopular and has yet to be tried (Read 2008).

It is clear that the continued incidental capture of large marine vertebrates in marine capture fisheries will further degrade the conservation status of a number of species and populations. Perhaps the biggest threat to these taxa, and one that is only at the infancy of being quantified is in small-scale artisanal fisheries. For marine mammals the ubiquitous use of gillnets in these fisheries is a serious concern. The scale of this problem and the socio-economic and political situations in much of the developing world where these fisheries operate require a comprehensive research programme to identify, test and implement successful bycatch mitigation strategies (Read 2008). In particular, there is a need for collaboration and knowledge transfer between scientists, gear technologists and social scientists working in this field. In this thesis, the study conducted in a small artisanal fishery in Argentina (Chapter 4) was part of an international project aiming to test the utility of stiffened or acoustically reflective gillnets at reducing small cetacean bycatch. Although these modified nets did reduce the bycatch of Franciscana dolphins in this trial, the project demonstrated how previous trials of bycatch mitigation devices could be built upon and tested in a collaborative manner in other fisheries. Such collaborative projects require substantial funding and there is a need to prioritise conservation resources and to maximise the utility of those experiments already conducted.

7.3   Future research

It is clear that further research is required to develop mitigation measures to reduce the bycatch of cetaceans in gillnets. While the use of pingers or time area closures has been shown to reduce bycatch rates, these strategies are expensive to monitor and enforce and only work under specific conditions. They are not a panacea for minimising bycatch in this type of fishing gear. The effects of rigged height and twine diameter as a potential gear modification to reduce the bycatch rates of cetaceans in gillnets should be explored further. However, the results of Chapter 4 indicate that for some species simply modifying gillnets may not prove a useful method of reducing bycatch. In such cases there is a need for the identification, development and testing of economically viable alternative fishing gears. Limited data are available on the behaviour of cetaceans interacting with fishing gear. An increased understanding of these interactions and the mechanisms that lead to entanglement would improve the ability of scientists, gear technologists and fishermen to develop bycatch mitigation strategies. Likewise, monitoring the behavioural reactions of non-target species to bycatch mitigation technologies is required to improve their design and continued success.

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