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Underwater Anthropogenic Sound:

Understanding the potential impacts on the marine environment and the influence on crab larval behaviour

Matthew Keith Pine

November 2013

A thesis submitted in fulfilment of the requirements for the degree of
Doctor of Philosophy in Marine Science.
The University of Auckland, NEW ZEALAND
Abstract

There is a need to identify and better understand the impacts of underwater anthropogenic sound on the marine environment. For example, very few investigations concerning anthropogenic sound impacts on crustaceans have been published. Therefore, the research described in this thesis characterises potentially significant anthropogenic sound sources in New Zealand’s coastal waters and begins to determine their influence on the settlement-stage larvae of estuarine crabs.

Several field-based experiments established that common sources of underwater anthropogenic sound in New Zealand were of appropriate intensities and frequencies to mask, over large areas, the natural underwater sounds found from several habitats. Laboratory-based experiments showed some anthropogenic sounds influenced the natural settling behaviour of settlement-stage larvae in several species of estuarine crabs. Notably, turbine sound was found to delay metamorphosis beyond a silent control. The research also characterised the ambient underwater acoustics from one of the largest estuaries in the world. Subsequent comparisons found that the sound from an operating tidal turbine was of greater intensity across the frequency range 0.1 – 20 kHz compared to the natural estuarine sound. The results also demonstrated that geometric spreading models are not accurate in shallow coastal waters and a simple model for the preliminary assessment of sound spreading was proposed based on field data.

The findings presented in this thesis extend our knowledge of the role of natural underwater sound in mediating larval settlement in important coastal organisms. Specifically, this work demonstrated that anthropogenic sound may mask natural underwater sounds in estuaries and is likely to influence the patterns of settlement of estuarine crab larvae. Overall, the results raise concerns about potential long-term ecological impacts of anthropogenic sound in coastal marine habitats.
Acknowledgements

This work cannot be attributed to just one individual, but to a team. I should firstly like to thank my supervisors, Professor Andrew Jeffs and Dr. Craig Radford, whose experience, time and patience confidently guided this work from concept to finished product. To Andrew, I say that your ability to guide the concepts from chaotic madness to published articles was truly outstanding and it has been an exceptional opportunity to learn from you. To Craig, I am equally grateful for your hard work, guidance and uncanny ability to conjure up new ideas from what may appear as ‘diddly-squat’. What I have learned from both of you I am sure I will carry far into the future.

I would like to extend gratitude to the University of Auckland’s Faculty of Science and OMV Ltd New Zealand for funding. I should also like to extend a massive thank you to Arthur Cozens, the operational manager at the Leigh Laboratory, for all your help in dealing with the money and resources. Your “just go for it and I’ll sort out the rest, mate” attitude was absolutely brilliant. Thank you. I also thank Dr. Chris Tindle for his help with the development of all equations and for the digital analogue of tidal turbine sound. I also thank Brady Doak, Peter Browne, and Errol Murray for all your help in the field and the workshop – it has been great and I know that every staff member and student at the Institute greatly appreciate what you do for us. I would also like to thank the laboratory’s electrical engineering genius, John Atkins, whose inventions from nuts and bolts, a few wires and bits of plastic, continue to blow my mind. I should also thank everyone else who helped me in the field, especially Olly, Charlie, Jenni, Soxi and Jess.

I would also like to thank my friends and family for all the love and support you have provided over the years. To mum and dad, I say a massive thank you to for always pushing me and never giving up. Your love and support is admirable and I could not ask for more.

And last but not least, a big thank you to my extraordinary Katie for looking after me while I’ve been dealing with this project over the last 3 years!
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Chapter 1 – General introduction

Chapter One

General introduction

1.1 Larval biology of Brachyuran crabs

New Zealand is inhabited by more than 78 species of Brachyuran crabs (true crabs) (Wear and Fielder 1985; Gunson 1993). The Cook Strait region is noted as containing the greatest diversity of these crabs in New Zealand, while in the Far-North regions of the country the diversity of Brachyura is at its lowest (Gunson 1993). Brachyuran crabs have lost their abdomen or ‘tail’ and thus are unable to swim as adults by using the pleopods under the abdomen as used by other decapods (Morton 2004). Their bodies are segmented and have ten legs. The carapace is flattened, is wider than long, and covers the head and thorax. With the legs being orientated sideways for running, the body is essentially rebuilt into a ‘wedge’ whereby they can fit into narrow spaces (Morton 2004). Their ecology is almost as diverse as they are; from shallow marine habitats such as the rocky intertidal, mud flats and sandy seafloors to living on seamounts and hydrothermal vents at depths of > 2500 m and even coping with a terrestrial existence in some species. Given the wide range of shore types in New Zealand there is a surprisingly low diversity of native crab species.

The larval development of most Brachyurans can be broken down into three parts, or phases (Wear & Fielder 1985). The first phase is the pre-zoea, or protozoea (Wear & Fielder 1985; Anderson 2001), and is very short-lived with moulting occurring within a few hours after hatching (Wear & Fielder 1985). Once the pre-zoea has moulted, it enters the second phase, which is known as the zoea (Wear & Fielder 1985; Anderson 2001; Miller & Harley 2007); a planktonic larva that swims with natatory setae on the exopods of maxillipeds 1 and 2 (Anderson 2001; Wear & Fielder 1985). Pre-zoea and zoea bear no morphological similarities to the adult. The duration of the zoea phase depends on the number of zoea stages (varies between species), ambient temperatures and food availability (Wear & Fielder 1985). The final phase is the megalopa; a robust and planktonic larva, that swims with the natatory
setae on the abdominal pleopods. Megalopae can detect light as well as acoustic stimuli (Anderson 2001).

Figure 1.2.1: Annotated diagram of a Brachyuran crustacean showing whole body. A = antenna. AB = abdomen (underneath). C = claw. CA = carapace. E = eye. SL = swimming leg. WL = walking leg. Figure taken directly from http://library.enaca.org/Health/FieldGuide/html/04.htm#
1.2 Acoustic detection in crustaceans and fishes

1.2.1 Underwater sound propagation

Sound is transmitted through water as longitudinal waves, also called compression waves. These waves consist of alternating pressure deviations from the equilibrium state, which cause localised regions of compression and rarefaction, and corresponding oscillation of the water molecules (Urick 1983; Moyle & Cech 2004). The frequency of sound relates to the number of repeating compressions, or waves, per second which in term determines the wavelength of the sound; the longer the wavelength, the lower the frequency \( f \) (i.e., \( f = \frac{\lambda}{c} \), where \( \lambda \) is the wavelength and \( c \) is the speed of sound in water \( 1500 \text{ m s}^{-1} \)). Attenuation is the loss of sound energy, or sound intensity, as it passes through a medium. In seawater lower sound frequencies have lower attenuation over a given distance compared to higher frequencies. For example, a 500 Hz sound wave only loses 1 dB of intensity over a distance of 100 km of transmission in seawater, which is much less than frequencies above 500 Hz (Ainslie & McColm 1998; François & Garrison 1982a, 1982b). The audibility of sound at a given distance from the source is a factor of not only the level of the sound, but also the transmission properties of the local environment which are determined by scattering (reflections) and attenuation (absorption) (Richardson et al. 1995). The propagation of sound in shallow water is different than in deep water because it has greater interaction with the sea surface and seafloor. Therefore, the depth of water column, sea state and seafloor composition can greatly influence the propagation of sound in the sea, especially in shallow coastal waters.

1.2.2 Hearing in crustaceans and fishes

The reception of underwater sound by crustaceans is not well understood (Simpson et al. 2011), however the reception of sound these organisms are generally classified within three groups: superficial receptor systems on the body surface, statocyst receptor systems and chordotonal organs (Budelmann 1992).

Superficial receptor systems are for the detection of water disturbances (Budelmann 1992). They are found all along the external body surface of many crustaceans (Breithaupt & Tautz...
1990) and consist of either a single cuticular hair, or a group of hairs. At the base of each hair there is a flexible basal joint which is mechanically attached to between one and four sensory cells (Vedel 1985; Budelmann 1992). The hairs can be bent by water movements and thus mechanically stimulate the sensory cells. They are sensitive to water displacements of between 0.05 and 300 Hz, depending on the species, and have a minimum receptor amplitude threshold of at least 0.2 µm water displacements (Goodall et al. 1990; Budelmann 1992). Cuticular hairs have been described in decapod crustacea and particularly in lobsters and crayfish (Budelmann 1992).

Statocyst receptor systems are primarily equilibrium receptor systems and gravity receptor systems (Budelmann 1992; Anderson 2001). In decapod crustacea, statocysts also include angular acceleration receptor systems or differential accelerometer (Budelmann 1992), which are analogous to the fish ear (Moyle & Cech 2004). Within statocysts are small stone-like structures called statoliths which are three times the density of sea water (Montgomery et al. 2006). They are surrounded by setae which are accompanied with mechanoreceptor cells (Anderson 2001; Popper et al. 2001; Montgomery et al. 2006) (Figure 1.1.2). Within crustacea, statocysts are similar in design, however, the location of them vary between the basal segment of the antennule and the uropods and telson (Budelmann 1992; Anderson 2001). When a sound pressure wave passes through the water, it displaces the water particles and any free-drifting materials (including larvae). As the statoliths do not move in synchrony with the seawater, due to the density differential, as this pressure wave passes, the statoliths brush against the setae which send signals through the antennule nerves (for decapods) to the suboesophageal ganglion (Anderson 2001). Evidence also suggests that statocysts of decapods are sensitive to substrate vibrations as well. In crayfish, statolith hair receptors clearly respond to both sound and vibration stimuli below 200 Hz (Budelmann 1992).

Chordotonal organs are widespread across crustacea and are generally associated with joints of flexible appendages (Budelmann 1992). In water, these appendages follow an oscillation caused by a sound wave in the seawater around it, whereby they stimulate the basal chordonal sensory cells.
Figure 1.2.2: (A) Diagram of a prawn showing whole body; (B) First antennule showing statocyst position; (C) The statocyst organ. Figure taken directly from Lovell et al. (2006).
Detection of environmental stimuli can vary between life cycle stages. For example, both in the nektonic swimming larva (puerulus) and benthic juvenile (post-puerulus) phases of *Jasus edwardsii*, pinnate sensory setae along the antennae have been suggested to be involved with vibration detection and thus perception of their environment (Jeffs et al. 1997).

The reception of underwater sound by fishes has two key components; the inner ear and the swim bladder (Moyle & Cech 2004). The inner ear also functions to orient or balance the fish within a three dimensional space, thereby proving the animal with a sense of direction even when suspended, neutrally buoyant or in a lightless environment (Moyle & Cech 2004). The inner ear is composed of semi-circular canals, the otolithic organs (the utriculus, sacculus and lagena; each containing an otolith) and sensory epithelia (Popper & Fay 1973) (Figure 1.2.3). The way in which the otolith functions in sound and gravity perception is analogous to the statolith within crustaceans. The sensitivity of hearing within the far field is significantly increased by anatomical adaptations that can transform sound pressures into displacement movements which create movement in the sacculus and lagena otoliths (Moyle & Cech 2004). Air within the fish’s swim bladder allows for this transformation and the compression of the gas within the bladder causes the surrounding tissues to vibrate; providing the necessary particle motion to cause otolithic stimulation. Some teleost species have a different system in which the swim bladder and auditory system is connected by a chain of small bones, called Weberian ossicles. Energy from the pulsating gas within the swim bladder is transferred down these ossicles to the auditory system (Moyle & Cech 2004).

The detection of underwater sound and its ecological relevance in true crabs is not well understood. To improve our understanding, and before the ecological implications and adaptive advantages to perceiving such environments can be introduced and further explored (as in this thesis), we must understand the natural ambient acoustics from all coastal environments.
Figure 1.2.3: Drawing of a medial view of the inner ear of (A) *Luciopera sandra* and; (B) *Cyprinus idus*. Modified from drawings by Retzius. Only the auditory portions of the ear are labelled. The nerves shown in both figures are the auditory portions of the 8th nerve. l – lagena; m – utriculus; o – otolith of each otolithic organ; s – sacculus; si – transverse canal. Figure taken directly from Popper and Fay (1973).
1.3 Under the waves is a noisy environment

A wide range of sounds characterise underwater environments that are generated by biological (biotic) and physical (abiotic) sources (Cato & McCauley 2002; Radford et al. 2010). A large majority of abiotic sounds underwater are due to the effect of wind interacting with the sea surface and waves, producing sound with dominant frequencies in the <10 – 1000 Hz range (Radford et al. 2008a, 2010). Biotic sounds are mainly attributed to soniferous animals engaged in reproductive and social behaviours, territorial defences and echolocation as well as incidental sound resulting from feeding activity and movement (Cato 1992; Radford et al. 2008a, 2010). These sounds are produced over a wide range of frequencies from below 20 Hz in fin and blue whales (Schevill et al. 1964; Cummings & Thompson 1971) to 200 kHz in dolphins and shrimps (Au et al. 1978; Cato and Bell 1992).

1.3.1 Temporal and spatial variation in the underwater sound environment

Several studies have shown patterns of periodic increases in the intensity of underwater biotic sound from coastal reefs which are referred to as dawn and evening choruses (Kaiser et al. 2005; Radford et al. 2008a,b). During any chorus, the intensity of ambient sound can increase by as much as 20 dB re 1 µPa² Hz⁻¹ which is due to the increased crepuscular activity of many reef inhabitants (Radford et al. 2008a). Furthermore, there can be variation within individual choruses which coincide with the lunar cycle (Radford et al. 2008a). For example, snapping shrimp showed significant lunar, diurnal and seasonal periodicity in their sound production, which accounted for the increases in the sound levels recorded in different habitats in the evening (Radford et al. 2010).

Nearshore environments may also be characterised by different underwater sounds. Marked differences have been found between the sounds emitted from three localised coastal habitats; a macroalgal-dominated reef, a sea urchin-dominated reef, and a sandy beach (Radford et al. 2010). Overall, the urchin-dominated reef produced significantly more intense sound in biologically relevant frequencies (800 – 2500 Hz) compared to the macroalgal-dominated and sandy beach habitats (Radford et al. 2010). There were also many differences in the sound among the habitats with the time of day the recordings were taken, showing that not only temporal compositions vary, but also the spectral composition between habitats.
1.4 The role of acoustics in the lives of crustaceans and fishes

Sound underwater has lower attenuation than in air meaning sound travels further underwater (Thomsen 2009). Light as an orientation cue for fish and crab larvae is only effective while it is detectable and it is therefore greatly limited due to its high attenuation in coastal waters. Chemical cues in aquatic environments are also limited by their detection distance. Therefore, underwater sound is considered as the principal cue for long-distance orientation from the open ocean to a desired settlement coastal habitat for fish and crab larvae (Radford et al. 2007; Stanley 2011). Thus, it is not surprising crustaceans and fishes have evolved hearing mechanisms that allow for the detection and cognition of underwater sound.

Our understanding of the detection of acoustic stimuli by pre-settlement crustacean and fish larvae has slowly improved over the last 20 years. The planktonic larvae of fish and brachyuran crustaceans must find a suitable reef habitat if they are to settle and grow (Simpson et al. 2005, 2011; Mann et al. 2007; Radford et al. 2007, 2008a, 2010; Stanley et al. 2010). There is increasing evidence that underwater sound cues are used by the larvae of fishes and crustaceans to orient themselves towards reef habitats and settle once they arrive at the source (Leis et al. 2003; Simpson et al. 2005, 2011; Montgomery et al. 2006; Mann et al. 2007; Radford et al. 2007, 2008b, 2010; Stanley et al. 2010). Underwater sound is thought to act as an important cue for settlement-stage larvae because it can travel long distances with minimal attenuation, whilst also conveying information regarding the quality of and direction to habitats (Rogers & Cox, 1988; Jeffs et al. 2003; Simpson et al. 2005; Radford et al. 2007, Simpson et al. 2011).

Jeffs et al. (2003) demonstrated first hand that larval crustaceans may orient toward underwater reef sounds by using light traps coupled with an artificial source of natural ambient sound. The results showed significantly greater numbers of larvae in light traps coupled with sound compared to silent traps, although this effect was only observed during particular moon phases and no effect was seen near full or new moons when tidal currents would be strongest. Other scientists have also reported strong evidence for the attraction of larvae to reef sound in five common New Zealand coastal crab species (Plagusia chabrus,
Notomithrax ursus, Cyclograpsus lavauxi, Hemigrapsus edwardsii and Pagurus spp.) (Radford et al. 2007).

Underwater sound has also been found to act as a settlement cue in both temperate (Hemigrapsus sexdentatus, Cyclograpsus lavauxi, Macrophthalmus hirtipes) and tropical (Grapsidae spp.) crab species (Stanley et al. 2010). Larvae subjected to reef sound showed a significantly shorter time to metamorphosis than individuals in the silent treatment, across all species. These results provide the first experimental evidence that underwater sound can advance the physiological development of larval decapod crustaceans.

Simpson et al. (2005) demonstrated that larval fish orient toward underwater reef sounds by building 24 reef patches from dead coral on sand flats and at each patch they deployed underwater loudspeakers which broadcasted reef sound, predominantly consisting of snapping shrimp and fish calls. The results showed greater fish diversity and abundance on reef patches which broadcast reef sounds, compared to those which did not. Also, Tolimieri et al. (2000) reported a median of 350 individual fish larvae entering light traps with sound (sound traps), compared to only 24 entering light traps without sound (silent traps). By using binary choice chambers with recorded reef sounds being played on one end, Tolimieri et al. (2004) found that significantly more triplefin and damselfish larvae were orienting towards the sound during the night – coinciding with similar findings from Leis et al. (1996) and Stobutzki and Bellwood (1998).

Despite the vast amount of literature describing the possibility of sound as an orientation and settlement cue, very few studies have attempted to identify the specific sound frequencies to which fish and crustaceans are responding. Simpson et al. (2005) found that higher frequency sounds (where 80% of the spectral energy was greater than 570 Hz – predominantly shrimp) attracted more fish taxa generally, compared to low frequencies (80% of spectral energy less than 570 Hz – predominantly produced by fish). What they also found was that pomacentrid (damselfish) larvae or juveniles were preferentially attracted to the higher frequency sounds, while apogonids (cardinalfish) were equally attracted to both high and low frequency acoustic signals. Thus, there is evidence that some fishes are discriminating between sound frequencies and are attracted to specific sounds (see also Radford et al. 2010). There is also evidence that some decapod crustaceans may be discriminating between sound frequencies.
with the mediation of settlement and metamorphosis to reef sound, rather than sound emanating from an estuary (Stanley et al. 2011).

1.5 Anthropogenic sound: from discovery to understanding

Anthropogenic sound is any sound generated by human activity (Slabbekoorn et al. 2010). Anthropogenic sounds which are of specific interest to this research are those which are within the audible frequencies of the receiver and are loud enough to overpower ambient sound levels (Thomsen 2009). In general terms, masking can be defined when “when a noise interferes with or obscures a signal” (Erbe 2008). For this research, masking of natural ambient sounds is considered to occur when the anthropogenic sound is louder than biologically important sounds and thus impairs the receiver’s ability to detect and assess the source in space and time (Thomsen 2009).

1.5.1 Underwater anthropogenic sound sources

Research investigating underwater anthropogenic sound has been increasing since the end of the First World War (Ross 1976). Anthropogenic sound is estimated to double in intensity every decade in coastal waters in some regions of the world with intense shipping activity (Wright 2008; Frisk 2012). The sources of anthropogenic sounds are wide-ranging and include ships, boats, seismic exploration devices (e.g., air guns), construction activities (e.g., pile driving) and sonar (Green & Moore 1995; Popper & Hastings 2009). Shipping and boat sound is a major anthropogenic sound source and can increase ambient levels within harbours and open oceans considerably (Popper & Hastings 2009). Motorized shipping has increased ambient sound levels at frequencies below 100 Hz in the deep sea by approximately 15 dB since the 1960’s (Ross 1993; Mazzuca 2001; Andrew et al. 2002). Most shipping sound is low frequency (< 300 Hz (Wright 2008)), however, sound from a modern cargo ship travelling at 16 knots can have much higher frequencies at intensities over 150 dB re 1 µPa at 1 m at 10 Hz, over 160 dB re 1 µPa at 1 m at 100 Hz and 180 dB re 1 µPa at 1 m at 200 – 500 Hz (Erbe 2012).

Industrial construction activity is a major source of underwater anthropogenic sound. Such activity includes pile-driving, dredging, drilling, installing offshore wind farms and blasts
from air guns and explosives (Popper & Hastings 2009a,b; Thomsen 2009) and can produce sound levels greater than 200 dB re 1 µPa @ 1 m (Erbe 2009a, 2009b, 2012). Pile-driving is increasingly common in coastal waters and can produce frequencies between 20 Hz to more than 20 kHz, with most energy reported between 100 Hz and 200 Hz (Madsen et al. 2006; Thomsen 2009). Marine dredging is used to deepen channels and harbours to mine seabed resources and it produces sound levels above 160 dB re 1 µPa @ 1 m with much energy between 50 and 500 Hz (Greene and Moore 1995; Robinson et al. 2011; Reine et al. 2012). Drill ships and semi-submersible drill rigs can produce sound levels of 191 (dB re 1 µPa @ 1 m at broadband frequencies (10 Hz – 10 kHz) (Greene 1987; Nedwell & Howell 2004). Explosions are often used during construction to remove subsurface structures and even in dredging when boulders are too large to be moved in one piece (Thomsen 2009). Explosions produce the highest sound level (274 dB re 1 µPa @ 1 m (Greene & Moore 1995)) from a point source in the sea with the ability to travel great distances (Richardson et al. 1995; Thomsen 2009).

As the demand for energy rises each year, offshore wind farms and tidal turbines are becoming more common, with most of them being built in shallow waters (<20 m) (Madsen et al. 2006). Underwater sound emanating from wind farms has two main sources; air flow through the wind blades and machinery sound (Tougaard et al. 2009), producing underwater sounds below 1 kHz (Fristedt et al. 2003; Lindell 2003; Wahlberg & Westerberg 2005) at 154 dB re 1 µPa @ 1 m at a wind speed of 13 m s\(^{-1}\) (Wahlberg & Westerberg 2005). Wind speeds, wind turbine size and the number of turbines affect underwater source levels and the distance at which fishes and marine mammals can hear them. Tidal turbines have been estimated to produce a source level of 175 dB re 1 µPa @ 1 m for frequencies between 200 and 8000 Hz (Parvin et al. 2005). Initial installation and turbine operation are the main sources of sound in tidal turbines.

Petroleum exploration is a source of high intensity sounds which can impact marine life. Such exploration can involve the repetitive use of high energy sound sources, such as airguns, which produce short, sharp low-frequency sounds (McCauley et al. 2003) and sound levels of up to 230 dB re 1 µPa @ 1 m (Pearson et al. 1994). Military sonar and seismic surveys have also shown to impact marine mammals (Parsons et al. 2008; Gordon et al. 2003).
1.5.2 Potential sound impacts on marine organisms

Over the last 20 years, there has been an increasing concern for the impact these sounds have on marine mammals, fishes and invertebrates. Currently, however, there is not enough data to predict how anthropogenic sound will alter ecosystems (Popper & Hastings 2009a). Thomsen (2009) describes several studies which have shown possible detrimental impacts from anthropogenic sound on marine organisms. Such impacts include disrupted communication among cetaceans and porpoises, as well as decreased abundance of cetaceans in areas of marine construction (such as wind farms, turbines, oil rigs, pile-driving and dredging) (Thomsen 2009; Erbe 2012).

In fishes, hearing loss and increased mortality has also been linked to high sound levels (Popper & Hastings 2009b), such as with the shiner surfperch (Thomsen 2009). Loud anthropogenic sounds induced stress responses and hearing loss in the goldfish *Carassius auratus* (Smith et al. 2004), while air-guns were found to severely (and evidence suggesting permanently) damage the hearing structures of fish (McCauley et al. 2003). High intensity sounds have been found to affect behavioural responses and act as a distraction to important acoustic signals, such as that given from a predator. For example, boat sound had a significant effect on the behaviour of the Caribbean hermit crab with simulated predators getting closer during sound playback experiments (Chan et al. 2010). Similarly, three-spined sticklebacks (*Gasterosteus aculeatus*) showed poorer foraging performance (measured by decreased discrimination between food and non-food items and food handling errors) in treatments exposed to white sound (bandwidth 100 – 1000 Hz) compared to the silent control (Purser & Radford 2011). Sound transmitted from boats has been found to mask communication signals between vocal fishes, such as *Chromis chromis*, *Sciaena umbra* and *Gobius cruentatus* (Codarin et al. 2009) and the Lusitanian toadfish, *Halobatrachus didactylus* (Vasconcelos et al. 2007), and disrupt the schooling behaviour of the blue fin tuna, *Thunnus thynnus* (Sará et al. 2007). Ship sound has also been found to increase the secretion of the stress hormone cortisol in freshwater fishes (Wysocki et al. 2006).

The impacts of offshore wind farms are more localised than other anthropogenic sound sources, with wind farm construction being of greatest concern (Madsen et al. 2006; Petersen & Malm 2006). Actual recordings of offshore wind farms are rare, and little is known about
their impacts on marine life. Sounds produced during the operation of a wind turbine were found to have no physiological impacts on fishes, harbour seals and porpoises (Wahlberg & Westerberg 2005; Tougaard et al. 2009). Even within 10 m of an operating wind turbine, the received levels were much lower than those required to cause temporary and permanent hearing damage to fishes (Wahlberg & Westerberg 2005), and have been described as incapable of masking communication between harbour seals and porpoises (Tougaard et al. 2009). However, some caution should be taken about interpreting potential impacts of wind farms on marine mammals and fishes as there are huge uncertainties surrounding the data on sound impacts (Wahlberg & Westerberg 2005) and thus, our understanding about offshore wind farms is poor.

Over the last 20 years there has been growing concern regarding the possible impacts of anthropogenic sound in the ocean. There is considerable evidence to suggest that anthropogenic underwater sound may impact many species and their behaviours (as in the Caribbean hermit crab and vocal fishes), yet currently, there is insufficient data available to support, or negate, the growing concerns that anthropogenic sound may change whole ecosystems. We need much more species- and habitat-specific data before conclusions can be drawn about how these sounds affect an ecosystem (Popper & Hastings 2009a).

1.6 Anthropogenic sound in the marine environment: the need for more research

The ocean is filled with underwater biological and physical sounds, many of which are important for the reproduction and survival of many fish and crustacean species (Slabbekoorn et al. 2010). While many impacts from anthropogenic underwater sound have been reported for marine mammals and adult fishes, there are very few studies which have dealt with larvae. Slabbekoorn et al. (2010) describe several hypotheses regarding the effects of increased sound levels in the ocean and emphasise some major targets for future bioacoustics research. For example, sound-induced stress has been seen in some fishes, however, we do not know if this stress could achieve a chronic status, in which case, growth and reproduction could be impaired (Slabbekoorn et al. 2010). Also, it is not well established whether or not every day anthropogenic sounds from vessels or cities impose possible masking effects, or detrimental impacts to shallow-water fishes and crustaceans. Furthermore, a lot is known about the importance and application of natural reef sound in mediating settlement in larval
crabs, however, very little is understood about how anthropogenic sounds may impair or inhibit such settlement. This could potentially lead to reduced recruitment to reef habitats, and in turn changes in long-term population dynamics.

There is a growing concern regarding the increase in underwater anthropogenic sound and our poor understanding of the ecological impacts, both short and long-term. With a growing appreciation of the ecological and economic importance of coastal fish crab species around the world, and the significant increase in anthropogenic sounds in coastal environments, a greater understanding of the sources of anthropogenic sound in New Zealand, the possible sound impacts on acoustically sensitive larvae, the ambient acoustic environment in shallow coastal habitats and sound propagation in shallow waters is essential.

1.7 Aims and thesis organisation

The growing concern and interest surrounding anthropogenic sound impacts was the key motivation of this research. No studies concerning the influence of anthropogenic sounds on the settlement behaviour on crabs have been published, and very few studies concerning how anthropogenic sounds propagate in shallow-waters of New Zealand are available. The overall aim was to explore a variety of anthropogenic sound sources and how such sounds may affect the settlement behaviour of crab larvae, using the larvae of several common estuarine crabs found throughout New Zealand. This was achieved using playback experiments which employ laboratory-based settlement tanks. Such methods have previously been used successfully to determine the responses of crab larvae to natural sources of underwater sound (Stanley et al. 2010). The specific aims of the thesis were:

1. Chapter Two: Analyses of potentially significant sources of anthropogenic underwater sound in New Zealand
   - Investigate a range of potentially significant anthropogenic underwater sound in New Zealand by determining the spectral composition and source levels of common sources of anthropogenic sound in New Zealand;
   - To provide a preliminary assessment of the potential to mask underwater acoustic signals by determining the distance at which any given anthropogenic sound may be detected by a generic fish or crustacean species.
2. Chapter Three: The influence of anthropogenic sound on the settlement behaviour in estuarine crab megalopae
   - To determine the metamorphosis response of the megalopae of two common estuarine crabs in New Zealand, *Austrohelice crassa* and *Hemigrapsus crenulatus*, to natural ambient estuarine sound;
   - To determine whether the underwater sound emitted from tidal and sea-based wind turbines influence the metamorphosis response of the crab megalopae;
   - Attempt to identify which characteristics of turbine sound are responsible for eliciting any observed changes in metamorphosis behaviour of the megalopae.

3. Chapter Four: Characterising ambient underwater sound of biological origin from a proposed tidal turbine site, the Kaipara Harbour
   - To investigate the ambient sound of biological origin from within two habitats (subtidal mudflat and seagrass-dominated habitat);
   - Provide baseline data for diurnal, lunar and seasonal variation in ambient sound within and between two estuarine habitats to better assess the potential for ambient sound masking by anthropogenic sound.

4. Chapter Five: The propagation of underwater anthropogenic sound in shallow water
   - Investigate the propagation of underwater tidal turbine sound from both a single (monopole) and double (twin monopole) sound source;
   - Assess the accuracy of widely used geometric spreading models by comparing them to field data and develop a more conservative and accurate spreading model for recommended use by regulatory bodies.

5. General Discussion
   - A general discussion of the findings in this thesis.
Chapter Two

Analyses of potentially significant sources of anthropogenic underwater sound in New Zealand

2.1 Introduction

Many studies have concluded that ambient underwater sounds are used as orientation and settlement cues by a broad range of species of larval reef fishes and crustaceans (Stobutzki & Bellwood 1998; Tolimieri et al. 2000; Jeffs et al. 2003, 2005; Leis & Lockett 2005; Simpson et al. 2005, Montgomery et al. 2006; Radford et al. 2007; Simpson et al. 2008a, 2008b, Stanley et al. 2010). For example, several species of crab post-larvae have been shown to swim toward sources of reef sound (Radford et al. 2007). Also, the time to metamorphosis (TTM) in crab post-larvae was found to markedly decrease in the presence of natural reef sounds when compared to silent controls (Stanley et al. 2010).

Natural ambient underwater sounds cover a wide range of frequencies from a diversity of sources, both abiotic and biotic (Cato & McCauley 2002; Radford et al. 2010). Underwater biological sounds produced from reefs, and other nearshore environments are within the audible frequency range of many fish larvae, and are of sufficient intensity to be detected by the larvae (Leis & Lockett 2005; Simpson et al. 2005; Wright et al. 2005; Montgomery et al. 2006; Simpson et al. 2008a; Wright et al. 2008). Hearing ranges vary between species, for example, the settlement-stage larvae of Pomacentrus nagasakiensis, P. amboinensis, Plectropomus leopardus and Lutjanus carponotatus all show sensitivity to frequencies below 2 kHz (Wright et al. 2010), while the American shad (Alosa sapidissima) can detect sounds from 100 Hz to 180 kHz, with two regions of greatest sensitivity, one from 200 to 800 Hz and the other from 25 kHz to 150 kHz (Mann et al. 1998).
Underwater anthropogenic sound is attributed to a wide variety of human activities, both in and around the sea. Major sources of anthropogenic sound in the oceans include construction and industrial activities, such as pile driving, seabed drilling, petroleum exploration and the use of sonar, as well as the movement of vessels (Hildebrand 2009). Motorways and causeways are also potential sources of anthropogenic sound which have been largely ignored in the literature. These anthropogenic sounds are of concern as they increase ambient sound levels within harbours and the open ocean (Popper & Hastings 2009). Some experimental studies have reported evidence for the detrimental impacts these sounds have on marine organisms, such as the disruption of communication among marine mammals (Thomsen 2009) and several species of fish (Vasconcelos et al. 2007; Codarin et al. 2009), hearing loss in the goldfish, Carassius auratus (Smith et al. 2004a), and chronic releases of stress hormones (Wysocki et al. 2006). One of the more prominent concerns with underwater anthropogenic sound pollution is acoustic masking (Slabbekoorn et al. 2010). Acoustic masking occurs when the anthropogenic sound is louder than natural ambient sound and thus impairs the receiver’s ability to detect and assess biologically important sounds (either communicative signals or environmental cues) in space and time (Thomsen 2009). Masking is also a function of an animal’s hearing threshold, and audible frequency range (Walhberg & Westerberg 2005).

Many underwater anthropogenic sounds are extremely loud in relation to ambient background sounds, and encompass frequencies within the audible frequency range of many fish and crustaceans – thus having the potential to mask ambient sounds (Slabbekoorn et al. 2010). Furthermore, many sources of anthropogenic underwater sound are located in migratory routes for cetaceans and oceanic fishes, and encompass both coastal and continental shelf waters which contain important habitats for crustaceans and fishes (Hildebrand 2009).

Sound intensity levels and sound frequencies vary depending on the type of anthropogenic activity and have been well studied. For example, seabed drilling produces sound levels of approximately 190 dB re 1 µPa @ 1 m at broadband frequencies (10 – 10,000 kHz) (Greene & Moore 1995), while offshore wind farms can produce 154 dB re 1 µPa @ 1 m at frequencies below 1 kHz (Wahlberg & Westerberg 2005). Thus, for any particular region or habitat, the nature of anthropogenic sound will differ and sounds need to be independently
analysed before any impact can be tested for any given marine organism in any particular region.

As the demand for renewable energy sources grows, offshore wind farms and underwater tidal turbines are becoming increasingly attractive (Gaudiosi 1999; Gill 2005; Bailey et al. 2010). Offshore wind farms are more common in Europe, yet their environmental impacts remain unclear (Wahlberg & Westerberg 2005). In New Zealand, a number of wind farms have been installed on land, but proposals for installation at sea are yet to be put forward. The Cook Straight would be an ideal location if offshore wind farms were to be proposed in the future because the wind is consistently high. However, underwater tidal turbines have been recently proposed for three locations in New Zealand, and local government has recently granted permission for the installation of such turbines in the Kaipara Harbour. To date, very little is known about the environmental effects of tidal turbines and the sound that they produce. The Kaipara Harbour is considered a significant nursery ground for several commercially important fish species, as well as hundreds of crustacean species (NIWA 2009; Paul 2009), and several fisheries could be threatened by the presence of tidal turbines.

There have been no published comprehensive studies of anthropogenic underwater sound sources in New Zealand. Also, few studies have been published to identify the sound transmitted from causeways and coastal motorways / highways, and only Dahl et al. (2007) have acknowledged and investigated their presence as a source of underwater anthropogenic sound. Therefore, the overall aim of the study was to investigate a range of potentially significant anthropogenic underwater sounds in New Zealand by determining the spectral composition and source levels of common sources of anthropogenic sound in New Zealand. Current anthropogenic sounds (commuter ferries, container ships and causeway sound), as well as potential sources such as offshore wind turbines and tidal turbines, were selected for investigation. Due to the absence of any local wind and tidal turbines in New Zealand, examples from the United Kingdom and Denmark were examined more closely and the sound generated by a mid-water (MW) tidal turbine and an offshore wind turbine were discussed in this study. To provide a preliminary assessment of the potential to mask underwater acoustic signals, the distance at which any given anthropogenic sound may be detected by a generic fish or crustacean species was also investigated.
2.2 Methods

2.2.1 Recording sites for vessels and causeway

Recordings of the sound emitted from the *Buxlink*, *Golden Bay* and *Maersk Batur* container vessels, the Research Vessel (R.V.) *Hawere*, and R.V. *Yellow Naiad*, the *Kawau Kat* and *Superflyte* ferries, the Royal New Zealand Navy vessel (*HMNZS Wellington*), and a motorway causeway were obtained from the field.

The sound emission from the largest Auckland commuter ferry, the *Superflyte* (aluminium catamaran, 38.8 m long, speed at time of recording 46 km h\(^{-1}\)), and the smaller *Kawau Kat* (20 m long, speed at time of recording 28 km h\(^{-1}\)), were recorded on 5 and 11 of August 2011, respectively, with good weather and sea conditions (19 km h\(^{-1}\) winds, 75 % cloud cover and 0.2 m swell). The recordings were obtained within the Rangitoto (36° 46.575' S 174° 48.656' E) and Motuhei Channels (36° 48.192' S 174° 54.817' E) (Figure 2.2.1b, c). Ambient control recordings were taken in the same location when no ferries or other vessels were visibly present or audible (confirmed using a calibrated hydrophone).

The sound emission from the University of Auckland’s R.V. *Hawere*, (MSA # 129774, 15 m long, speed 28 km h\(^{-1}\)) was recorded outside Leigh Harbour (36° 18.16' S 174° 49.04' E) on 14 December 2010, with good weather conditions (Figure 2.2.1a). The University of Auckland’s R.V. *Yellow Naiad*, (MSA # 132907, 4.8 m long, aluminium hull, inflatable beam, speed 43 km h\(^{-1}\) at time of recording) was also recorded outside Leigh Harbour (36° 18.305' S 174° 48.602' E) on 5 of August 2011, with slight sea conditions (0.5m swell, 27 km h\(^{-1}\) SW winds) (Figure 2.2.1a). Ambient control recordings were taken in the same location in the absence of the respective anthropogenic source on the same day.

The sound from the *Golden Bay* (IMO # 7720257, 98 m long, 4,493 t, average speed 26 km h\(^{-1}\)) and *Maersk Batur* (IMO # 9402029, 223 m long, 35,835 t, average speed 26 km h\(^{-1}\)) container vessels were recorded in the Rangitoto Channel (36° 46.575' S 174° 48.656' E) on 11 August 2011 in good weather conditions (15 km h\(^{-1}\) winds, 0.5 m swell, 50 % cloud cover) (Figure 2.2.1b). The sound from the *Buxlink* container vessel (IMO # 9235816, 206 m long, 33,817 tonnes, average speed 29 km h\(^{-1}\)) was recorded in the Waitemata Harbour, while the
ship approached its berth, in good sea conditions (20 km h\(^{-1}\) winds, 0.5 m swell, 90% cloud cover) (Figure 2.2.1d) on the 9 of August 2011.

The sound from the offshore navy patrol vessel, the *HMNZS Wellington* (85 m long, 1900 t, 19 km h\(^{-1}\)), was recorded in the Rangitoto Channel (36° 46.596' S 174° 48.706' E) in good sea conditions (15 km h\(^{-1}\) winds, 0.5 m swell, 50% cloud cover) on the 11 August 2011 (Figure 2.2.1b).

The sound emission from a local six lane causeway (State Highway 16) (36° 52.15' S, 174° 51.34' E) (Figure 2.2.1e) was recorded over four days, on the hour for 15 minutes, (2 - 5 of February 2011), with good weather conditions (19 km h\(^{-1}\) winds, 60% cloud cover and no swell). Ambient control recordings were taken 2.3 km from the causeway (36° 51.05' S, 174° 41.49' E) on the same dates at the same time as the motorway recordings.

### 2.2.2 Recording systems

Recordings of commuter ferries, container ships and commercial vessels were collected from the field using a calibrated omnidirectional hydrophone (HTI-96-MIN, High Tech Inc., USA) connected to a watertight recording housing unit (set at 20 dB gain, 16 bit, 48 kHz sampling rate) (Figure 2.2.2b). The recording unit was free floating during ideal conditions to reduce any extraneous sound associated with recording off a vessel. Recordings (digitalised to a .WAV file) of vessel sound lasted for 10 min while each sound source was visible and ambient recordings were 15 min long while the source vessel or other vessels were not in view or audible using a calibrated hydrophone.

Sound transmitted from a local causeway was collected using a calibrated High Tech, Inc. HTI-96-MIN omnidirectional hydrophone (10 Hz to 60 kHz flat response) connected to a watertight temporal recording unit (Figure 2.2.2a). The recording apparatus (containing the hydrophone, battery, recorder (20 dB gain, 16 bit, 48 kHz sampling rate) and timer) was bolted to a steel stand anchored by an iron bar and submerged in 3 m of water at mean low water. The marker float was anchored 10 m from the recording apparatus to remove any extraneous sound from the float or rope. A digital timer was used to activate the recorder for 15 min on the hour of every hour for four days.
Figure 2.2.1: Map showing the location of recordings for: A) R.V. Hawere and R.V Yellow Naiad; B) The HMNZS Wellington Naval vessel, the Golden Bay and Maersk Butar container vessels, and the Kauau Kat ferry; C) The Superflyte ferry; D) The Buxlink container vessel; E) Motorway causeway; F) The proposed location for tidal turbines in the Kaipara Harbour.
Figure 2.2.2: Schematic diagrams of recording systems: A) temporal recording unit used for the causeway; B) Free floating underwater recording unit used for the recording of vessels (either attached to a stationary vessel, or drifting).
Digital recorders (EDIROL-R09HR) were calibrated before each recording. Recordings (digitised to a .WAV file recording) were made using a fixed gain level of 20 dB on the recorder. The recorder was calibrated to this setting by taking a series of 1 kHz pure tone sine wave recordings produced by a signal generator (Incorporate Electronics Corporation, model F34 function generator). The voltage ($V_{rms}$) of the recorded 1 kHz tone was measured and compared with the hydrophone sensitivity of -164.6 dB re 1 V / 1 µPa. From the data, a correction factor was calculated and used within MATLAB with codes specifically written for these recordings for sound pressure measurements. Hydrophones were field calibrated before and after each recording episode using a piston phone.

2.2.3 Data analyses

Sound intensity spectra plots for five random 10 sec section of each recording, when the vessel was closest to the hydrophone, were generated to compare anthropogenic sound sources with background sound levels. The sound intensity spectra for each anthropogenic sound were generated using MATLAB acoustic software. Recording data (both anthropogenic sound and ambient recordings) was high-pass filtered at 100 Hz to reduce interference by wind and surface waves in shallow water, which typically transmit frequencies below 50 Hz (Stanley 2011). The total sound intensity ($Prms^2$) was calculated for each 10 sec anthropogenic sound sample and corresponding control (ambient) sample. Ambient recordings (controls) were made in the same location once the source was absent to distinguish the anthropogenic sound from the ambient sound.

Each 10 sec anthropogenic sound and control recording was band pass filtered into seven different frequency groups: 100 – 400, 401 – 800, 801 – 1200, 1201 – 2000, 2001 – 5000, 5001 – 10000, and 10001 – 22000 Hz. These frequency groups were selected as most acoustic energy of anthropogenic sound resides in the lower frequencies (Popper & Hastings 2009) and also because many fishes and crustaceans show physiological and behavioural responses at frequencies below 5 kHz. For each frequency band, the sound intensities ($Prms^2$) were calculated, and used to evaluate the proportion of total sound intensity attributable to each frequency band for each sound source recording. Proportions of total sound intensities between and within anthropogenic and ambient sound recordings were analysed using one
way ANOVA, after the data was arcsine transformed and conformed to the assumptions of normality and homogeneity (Sokal & Rohlf 1995). Significant differences between individual frequency band pairs of means were determined using Holm-Sidak tests once an overall significant difference amongst means was determined from ANOVA.

2.2.4 Calculating source levels and detection distances

Source levels (SL) were back-calculated using the received level (RL) and transmission loss (TL). Transmission loss is the reduction of sound intensity as it travels through the water, and is defined by:

\[ TL(r) = SL - RL(r); \]  

(Eq. 1)

where \( r \) is the distance between the receiving hydrophone and anthropogenic sound source (Wahlberg & Westerberg 2005). Transmission loss occurs through spherical and/or cylindrical geometric spreading (the “dilution” of sound energy as it radiates out beyond the source (Wahlberg & Westerberg 2005)). Spherical spreading (given by 20 log \((r)\)) occurs in deeper waters, where the distance between the receiver and source is less than the depth; while cylindrical spreading (given by 10 log \((r)\)) occurs in shallower water where the distance between the receiver and source is greater than the depth (Wahlberg & Westerberg 2005). In the current study, all recordings were made in intermediate ranges, where the depth is less than the distance between the receiver and source. Therefore, \( TL \) was calculated using a modified theoretical model of underwater sound propagation by Richardson et al. (1995):

\[ TL = 20 \log RO + 10 \log R/RO + 0.04 + 10 \log \frac{d}{dO} + \alpha; \]  

(Eq. 2)

where \( RO \) is the range in metres before transition to cylindrical spreading, \( R \) is the range between the hydrophone and source in metres, \( d \) is the depth at the receiving hydrophone in metres, \( dO \) is the depth at the source vessel or turbine in metres and \( \alpha \) is frequency dependant, volumetric absorption (François and Garrison 1982a,b). An important assumption to note is that spherical spreading is assumed to occur to three water depths, after which cylindrical spreading is assumed. This is because bottom reflection is weak to approximately two water depths as silt and sandy seafloors typically have grazing angles of 20° (Jensen et al. 2011). The distance between each recording system and target vessel was estimated using a
measurement function on a GPS unit and several waypoints (5 m error estimate) to map the path of the travelling vessel (Figure 2.2.3). Depth and distance values for the tidal and wind turbines were estimated from hydrographical charts of the Kaipara Harbour (where tidal turbine installation is proposed) and the Cook Straight (where offshore wind turbines would likely be proposed). The $TL$ for a MW tidal turbine was estimated using a depth value of 32 m at the turbine and 30 m at the receiver (100 m from the turbine). The $TL$ for a wind turbine was estimated using a depth value of 45 m at the turbine and 120 m at the receiver (100 m from source). A bottom attenuation coefficient of 0.04 dB km$^{-1}$ was assumed (Radford et al. 2011).

Detection distances were calculated using the sonar equation:

$$SE = SNR - DT; \quad (Eq. 3)$$

where $SE$ is signal excess, $SNR$ is the signal to noise ratio, and $DT$ is the detection threshold (Clark et al. 2009). A detection distance was defined as the distance at which the sound emitted from a source is loud enough to be detected by a generic crustacean or fish species whose detection threshold is equal to or less than 6 dB above the lowest ambient background levels recorded (the 5th percentile) for 0.1 – 5 kHz. The prawn, *Palaemon serratus*, has been found to be acoustically receptive to frequencies up to 5 kHz at pressure levels greater than 131 dB re 1 µPa (5 dB higher than the measured broadband background sound for the inner Hauraki Gulf, and thus an assumed detection threshold of 5 dB above background sound levels) (Lovell et al. 2005). To the author’s knowledge this is the only study investigating the hearing thresholds of a crustacean, and therefore, a conservative detection threshold of 6 dB above the lowest (5th percentile) background sound levels for 0.1 – 5 kHz was assumed in the current study. Therefore, the anthropogenic sound must be at least 110 dB or 96 dB re 1 µPa @ 0.1 – 5 kHz in order to be detected by a generic fish or crustacean species in the inner and outer Hauraki Gulf, respectively. Conservative hearing thresholds of 6 dB above background sound levels have also been assumed for a generic fish species (Radford et al. 2005). Due to natural fluctuations of ambient sound levels and other features of sound, a signal may not be detected by an animal, even when the signal itself is louder than the ambient level (Clark et al. 2009). The difference between the ambient noise level and the signal level (noise-to-signal ratio) at which an animal can detect the signal is termed a detection threshold ($DT$). Signal
excess \((SE)\) is the relation between \(DT\) and ambient sound levels (without the targeted anthropogenic sound source) \((NL)\) and is defined as the 50\% probably of detection if the signal’s presence is random in space and time (Clark et al. 2009). Thus, the modified equation which was used for the calculation of detection distances was:

\[
SE = SL - NL - TL - 6;
\]  
(Eq. 4)

where \(SE\) equals zero. Detection distances were subject to three important assumptions: (1) the sound and detecting receiver are omnidirectional; (2) detection threshold is 6 dB above ambient background sound; and (3) ambient background sound (without the anthropogenic sound source) is 110 dB re 1 µPa @ 0.1 – 5 kHz (the ambient sound level during a summer’s day in the inner Hauraki Gulf) or 96 dB re 1 µPa @ 0.1 – 5 kHz (ambient sound level during a summer’s day in the outer Hauraki Gulf) or 94 dB re 1 µPa @ 0.1 – 5 kHz (an assumed hypothetical ambient sound level during a winter’s day in the Kaipara Harbour or Cook Straight). It is important to note that recordings of a particular anthropogenic sound source included the ambient sound.

Due to the absence of operational tidal and wind turbines in New Zealand only published power spectrums and source levels were used for the purposes of comparison with other anthropogenic sound sources (Parvin et al. 2005 (tidal turbine); Walhberg & Westerberg 2005 (offshore wind turbine)). The published source levels were used to estimate detection distances (using the same sonar equation as for vessels) for both the tidal and offshore wind turbines for the Kaipara Harbour and Cook Straight, respectively.
Figure 2.2.3: Schematic diagrams of how distance between receiver and source was measured for vessels: A) Method for measuring the distance from hydrophone for the R.V. Hawere and R.V. Yellow Naiad. Individual waypoints were created on the GPS unit which were followed at a set speed as source circumnavigates the hydrophone; B) Method for measuring distance ($r$) from vessels that were not under the control of the University of Auckland. Waypoints were created in the GPS at the hydrophone position and in the wake of the vessel once it had passed.
2.3 Results

2.3.1 Background intensity levels

Broadband background sound in the inner and outer Hauraki Gulf was 126 dB re 1 µPa and 109 dB re 1 µPa, respectively. A broadband background sound level of 105 dB re 1 µPa was assumed for the Kaipara Harbour where installation of 200 tidal turbines is proposed. A conservative broadband background sound level of 105 dB re 1 µPa was also assumed for a hypothetical location within the Cook Straight.

2.3.2 Spectral levels

The sound recorded from a motorway causeway showed a gradual peak between 100 – 400 Hz at approximately 90 dB re 1 µPa Hz$^{-1}$. Another peak was observed at 1 kHz at approximately 105 dB re 1 µPa Hz$^{-1}$; however this peak was also visible in the ambient control recording (Figure 2.3.1.e). The sound from the Superflyte was also of low frequency (80 – 1200 Hz) and showed a sharp peak at 1 kHz at approximately 130 dB re 1 µPa Hz$^{-1}$ (Figure 2.3.1a), similar to the R.V Hawere (70 – 1100 Hz, sharp peaks at approximately 2 kHz and 5 kHz), and the Kawau Kat ferry (60 – 1200 Hz, peaking between approximately 80 – 100, 200 – 400, and 500 Hz) (Figure 2.3.1i). The small R.V. Yellow Naiad showed high intensity levels across a wider frequency range (200 – 5000 Hz, multiple peaks between 150 – 1000, 200 and 5000 Hz) (Figure 2.3.1b). Frequencies below 2 kHz characterised all container vessels, with lower power being produced in the slower Buxlink container ship which was travelling approximately 11 km h$^{-1}$ at the time of recording. A sharp peak at approximately 1 kHz and gradual peak below 150 Hz was observed in the sound emitted from the Golden Bay container vessel (while travelling at 24 km h$^{-1}$) (Figure 2.3.1f). The Maersk Batur showed peaks in the spectrum at approximately 5000, 2000, 900 – 1100, 800, 500 and below 100 Hz (Figure 2.3.1g). The HMNZS Wellington also produced low frequency sound dominated by frequencies below 1 kHz. All peaks observed in the vessel sound spectrum were absent in the ambient control recordings. Wind turbine sound was again dominated by frequencies below 1 kHz, with peaks at approximately 50, 80 and 200 Hz, while underwater tidal turbines showed high intensity levels across a wider frequency range.
(200 – 5000 Hz) (Figure 2.3.2). Peaks in the tidal turbine spectrum were observed at frequencies 350, 1500, and 5000 Hz.

2.3.3 Total received sound intensity proportions

Frequencies below 1200 Hz made up 69.8 % ± 9.5 % of total sound intensity (acoustic energy) of the Superflyte, (Figure 2.3.3). This significantly differed from the smaller passenger ferry, the Kawau Kat, which comprised of 47.0 % ± 3.7 % of acoustic energy residing in frequencies below 1200 Hz ($P < 0.001$). The mean proportions of total sound intensity in each frequency band was significantly different for the sound emitted from the Kawau Kat ($F_{6, 0.05} = 4.343$, $P < 0.01$), Superflyte ($F_{6, 0.05} = 35.09$, $P < 0.001$), R.V. Yellow Naiad ($F_{6, 0.05} = 49.12$, $P < 0.001$), R.V. Hawere ($F_{6, 0.05} = 54.37$, $P < 0.001$), container vessels Golden Bay ($F_{6, 0.05} = 956.98$, $P < 0.001$), Maersk Batur ($F_{6, 0.05} = 40.16$, $P < 0.001$), and Buxlink ($F_{6, 0.05} = 87.45$, $P < 0.001$), HMNZS Wellington ($F_{6, 0.05} = 274.72$, $P < 0.001$) and motorway causeway ($F_{6, 0.05} = 137.29$, $P < 0.001$). The sound emitted from the R.V. Hawere, had 40.1 % ± 3.3 % of acoustic energy in frequencies below 1200 Hz and the R.V. Yellow Naiad had a significantly greater proportion of acoustic energy (46.0 % ± 4.2 %) in frequencies between 2001 and 5000 Hz compared to any other frequency band ($P < 0.05$). The fastest moving container vessel investigated in this study (travelling at 24 km h$^{-1}$ at time of recording), the Golden Bay, was characterised by 95.0 % ± 3.4 % of the acoustic energy in frequencies below 1200 Hz. The Maersk Batur container vessel (travelling at 15 km h$^{-1}$ at time of recording) showed 64.2 % ± 6.0 % of the acoustic energy in frequencies below 1200 Hz, and the berthing Buxlink container ship had 36.0 % ± 1.4 % of the acoustic energy in frequencies below 1200 Hz. The HMNZS Wellington was also dominated by frequencies below 1200 Hz (58.0 % ± 2.0 % total sound intensity). The acoustic energy within the frequency band of 100 – 400 Hz was significantly greater than any other frequency band in the Golden Bay ($P < 0.001$), Maersk Batur ($P < 0.001$) container vessels, the HMNZS Wellington ($P < 0.001$) and the Superflyte ferry ($P < 0.001$).

While frequencies below 1200 Hz made up 50.1 % ± 4.8 % of the sound emitted from the motorway causeway, only 3.5 % ± 0.4 % of the sound intensity was attributable to the frequency range of 100 – 400 Hz, which was significantly higher than the 2.15 % ± 0.02 % attributable in the ambient recording ($P < 0.05$). Most acoustic energy (45.0 % ± 3.8 %) from
the motorway causeway resided in the frequency band 801 – 1200 Hz ($P < 0.001$). Higher frequencies (> 5001 Hz) made up less than 30 % for all sources of sound, except for the *Hawere*. All ambient recordings had significantly greater proportions of total sound intensities in frequency bands above 5000 Hz ($P < 0.05$), while vessel and motorway sound showed greatest proportions of total sound intensities in frequency bands below 5000 Hz when compared to background sound levels ($P < 0.05$).
Figure 2.3.1: Received anthropogenic sound spectra: A) Commuter Ferry *Superflyte*; B) University of Auckland’s R.V. *Yellow Naiad*; C) University of Auckland’s R.V *Hawere*; D) *Buxlink* container vessel; (E) Motorway causeway; F) *Golden Bay* container vessel; G) *Maersk Batur* container vessel; H) *HMNZS Wellington* Naval vessel; I) *Kawau Kat* passenger ferry. The blue line represents anthropogenic sound recording, black line represents ambient recording in the absence of anthropogenic sound.
Figure 2.3.2: Anthropogenic source sound spectra: A) Several European offshore wind turbines (taken directly from Walhberg & Westerberg 2005); B) Tidal turbine from Lynmouth (taken directly from Parvin et al. 2005).
Figure 2.3.3: Proportion of total sound intensity (Prms$^2$) for nine different sources of anthropogenic noise in the Hauraki Gulf. A) the Superflyte; B) the Kawau Kat; C) Maersk Batur container vessel; D) HMNZS Wellington; E) R.V. Hawere; F) R.V. Yellow Naiad; G) Golden Bay container vessel; H) Buxlink container vessel; I) Motorway causeway.
2.3.4 Source levels and detection distances

A 6 dB above ambient sound level hearing threshold was assumed for a generic fish/crustacean species and audible frequency ranges below 5 kHz. Therefore, the sound at frequencies 0.1 – 5 kHz emitted from any anthropogenic source would need to be at least 116 dB re 1 µPa or 102 dB re 1 µPa or 100 dB re 1 µPa before being detected by a generic fish/crustacean species residing in the inner or outer Hauraki Gulf or the Kaipara Harbour and Cook Straight, respectively.

Detection distances varied depending on the source spectrum and the propagation environment, rather than source level alone (Table 2.3.1; Figure 2.3.4). The sounds emitted from all vessels in this study were estimated to have SL’s greater than 159 dB re 1 µPa @ 1 m. The sound emitted from the R.V. Hawere (SL of 176 dB re 1 µPa @ 1 m), travelled 0.94 km and 5.34 km before becoming ‘undetectable’ in the inner and outer Hauraki Gulf, respectively. The sound emitted from the Superflyte ferry had an estimated SL of 176 dB re 1 µPa @ 1 m and would have to travel a distance of 0.92 km or 8.94 km before becoming undetectable in the inner and outer Hauraki Gulf, respectively. Sound propagating from the Golden Bay container ship (a SL of 184 dB re 1 µPa @ 1 m) travelled the greatest estimated distance of 1.46 km and 31.68 km before becoming undetectable in the inner and outer Hauraki Gulf, respectively. The sound emitted from the R.V. Yellow Naiad (SL of 180 dB re 1 µPa @ 1 m) travelled 0.96 km and 11.84 km before becoming undetectable in the inner and outer Hauraki Gulf, respectively. The sound from most of the vessels in this study were estimated to travel more than 5 km before becoming undetectable in the outer Hauraki Gulf, with the exception of the sound emitted from the HMNZS Wellington (SL of 159 dB re 1 µPa @ 1 m and an estimated audible distance of 3.26 km), and the Kawau Kat ferry (SL of 170 dB re 1 µPa @ 1 m and an estimated audible distance of 2.66 km). The sound of a local causeway, however, had an estimated source level of 125 dB re 1 µPa @ 1 m and a detection distance of 0 m for both the inner and outer Hauraki Gulf.

The sound from the tidal turbines was estimated to be detected by a generic fish or crustacean within 2.1 km from the turbine structure (SL of 175 dB re 1 µPa @ 1 m (Parvin et al., 2005) in the Kaipara Harbour. The sound from an offshore wind turbine had the lowest SL of 151 dB re 1 µPa @ 1 m and was estimated to be detected within a distance of only 46 m in a
hypothetical location within the Cook Straight (assuming the same ambient sound level as the Kaipara Harbour of 105 dB re 1 µPa).
Table 2.3.1:  Source levels (dB re 1 µPa @ 1 m) and detection distances for several anthropogenic sound sources. Detection distances are for a generic fish/crustacean species and are based on a hearing threshold of 6 dB above ambient sound levels and an audible frequency range below 5 kHz.

<table>
<thead>
<tr>
<th>Sound Source</th>
<th>Distance from source (m)</th>
<th>Speed of vessel (km/h)</th>
<th>Average Depth (m)</th>
<th>RL (dB re 1 µPa)</th>
<th>SL (dB re 1 µPa @ 1 m)</th>
<th>Transmission Loss (dB)</th>
<th>Detection Distance (m) (@ 0.1 – 5 kHz)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Superflyte ferry</em></td>
<td>192</td>
<td>46</td>
<td>9.5</td>
<td>146</td>
<td>176</td>
<td>30</td>
<td>920***</td>
</tr>
<tr>
<td><em>Kawau Kat ferry</em></td>
<td>398</td>
<td>28</td>
<td>12.5</td>
<td>136</td>
<td>170</td>
<td>34</td>
<td>1030***</td>
</tr>
<tr>
<td><em>R.V. Yellow Naiad</em></td>
<td>136</td>
<td>43</td>
<td>21.5</td>
<td>149</td>
<td>180</td>
<td>31</td>
<td>960***</td>
</tr>
<tr>
<td><em>R.V. Hawere</em></td>
<td>200</td>
<td>28</td>
<td>29</td>
<td>144</td>
<td>176</td>
<td>32</td>
<td>940***</td>
</tr>
<tr>
<td><em>Buxlink container vessel</em></td>
<td>457</td>
<td>11</td>
<td>8.5</td>
<td>133</td>
<td>160</td>
<td>27</td>
<td>840***</td>
</tr>
<tr>
<td><em>Golden Bay container vessel</em></td>
<td>93</td>
<td>24</td>
<td>12.5</td>
<td>156</td>
<td>184</td>
<td>36</td>
<td>1460***</td>
</tr>
<tr>
<td><em>Maersk Batur container vessel</em></td>
<td>290</td>
<td>15</td>
<td>12</td>
<td>146</td>
<td>175</td>
<td>28</td>
<td>1330***</td>
</tr>
<tr>
<td><em>P140 Naval vessel</em></td>
<td>300</td>
<td>15</td>
<td>12.5</td>
<td>132</td>
<td>159</td>
<td>27</td>
<td>783***</td>
</tr>
<tr>
<td>Motorway causeway</td>
<td>20</td>
<td>-</td>
<td>3</td>
<td>104</td>
<td>125</td>
<td>21</td>
<td>0***</td>
</tr>
<tr>
<td>Wind turbine</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>151*</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Tidal turbine</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>175**</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

*Value obtained from Walhberg and Westerberg, 2005  
**Value obtained from Parvin et al., 2005  
***Inner Hauraki Gulf, where ambient level 110 dB re 1 µPa @ 0.1 – 5 kHz  
****Outer Hauraki Gulf, where ambient level 96 dB re 1 µPa @ 0.1 – 5 kHz  
*****Kaipara Harbour, where ambient level is assumed to be 94 dB re 1 µPa @ 0.1 – 5 kHz  
******Cook Straight, where ambient level is assumed to be 94 dB re 1 µPa @ 0.1 – 5 kHz
Figure 2.3.4: Log transformations for: (A) detection distances for a generic fish/crustacean species with 6 dB above background noise levels hearing thresholds and audible frequency range below 5 kHz, for eight different vessels; and (B) estimated transmission loss (TL).
Figure 2.3.5: Log transformation for estimated transmission loss (TL) for three different frequencies with distance along the horizontal.
2.4 Discussion

Vessel sound in this study showed most acoustic energy residing in frequencies below 5 kHz. Ambient sound from the inner Hauraki Gulf showed most acoustic energy residing in frequencies above 2 kHz, and was largely attributable to snapping shrimp. Sound from the offshore wind turbine and MW tidal turbine differed in spectral composition to vessels, with nearly all energy residing in frequencies below 1 kHz in the wind turbine and below 8 kHz for the tidal turbine. The sound source levels of all eight operating vessels were estimated to be very high, with all being greater than 158 dB re 1 µPa @ 1 m (Table 2.3.1).

With the majority of the acoustic energy residing in the frequencies below 2 kHz and of high source levels, anthropogenic sounds are capable of travelling significant distances. The engines, gearboxes and other operating machinery generate the sound that then passes through the hull into the water column (Hildebrand 2009). Cavitation at the propeller blade has also been found to be a considerable source of underwater sound across all frequencies, with the collapse of cavitation bubbles producing higher frequencies and the blade passage through the water producing lower harmonics (Hildebrand 2009). The interaction between the hull and the sea surface is another source of sound (Hildebrand 2009). Vessel sound varies with vessel load, speed, size and hull material (Table 2.4.1) (Hildebrand 2009). For example, the speed of container ships appears to influence the distribution of acoustic energy emitted (if vessel specifications were to be excluded), and engine type can produce different source levels (Table 2.4.1). The low frequency sound from a wind turbine is produced by the rotation of the aerial blades which is transmitted down the pylon structure and into the water column in which it stands (Wahlberg & Westerberg 2005). Tidal turbine sound comes from the mechanical gearbox, producing the peaks in the spectrum at 350, 1500, and 5000 Hz (Parvin et al. 2005) (Figure 2.3.1g). Turbines that have a hydraulic gearbox purportedly generate less sound (Parvin et al. 2005). The sound from wind turbines depends on wind speed and size; similarly with tidal turbines and current velocities (Table 2.4.1) (Hildebrand 2009).

The potential of anthropogenic sound to mask acoustic communication among fish is thought to be high (Slabbekoorn et al. 2010). While many studies have investigated the potential for sound to mask the acoustic communication and acoustic environmental cues for fish (see
Slabbekoorn et al. (2010), crustaceans and other invertebrates have been largely neglected. This has also meant that hearing and detection thresholds for Brachyuran crustaceans are largely unknown, and thus conclusions regarding the extent of masking biologically important sounds cannot be accurately inferred for any given species. Therefore, for the purpose of the current study, masking is assumed to not occur in the Hauraki Gulf when the vessel sound can no longer be detected by a given fish or crustacean species. Conclusions based on estimations which exclude detection thresholds may overestimate true detection distances. Thus, a detection threshold of 6 dB above ambient sound levels for 0.1 – 5 kHz has been assumed and thus care should be taken when reviewing detectable distances for each sound source. The threshold of 6 dB is conservative, and if one was to alter the detection criteria to equal levels of anthropogenic sound and the ambient sound, the detection distances would more than quadruple (6 dB is a factor of 4 in intensity) (Radford et al. 2005).

All anthropogenic sources examined in this study, excluding the motorway causeway, have the potential to mask natural ambient underwater sound due to their intensity and spectral composition. While hearing thresholds for decapod crustaceans are unknown, earlier research has shown the statocysts of the prawn, *Palaemon serratus*, to be receptive to low-frequency sounds (100 – 3000 Hz) (Lovell et al. 2005). As this is the only study to examine the hearing thresholds of crustaceans, it is difficult to infer these thresholds to decapod crabs. However, many fish species have shown auditory responses to low-frequency anthropogenic sound, for example, the Lusitanian toadfish (*Halobatrachus didactylus*) (Vasconcelos et al. 2007), bluefin tuna (*Thunnus thynnus*) (Sarà et al. 2007), brown meagre (*Sciaena umbra*), red-mouthed goby (*Gobius cruentatus*) and the damselfish (*Chromis chromis*) (Codarin et al. 2009). Furthermore, reef systems produce sounds with highest intensities in frequencies below 5000 Hz. For example, an evening chorus at a temperate reef was within the frequency band of approximately 400 to 5000 Hz (Radford et al. 2008a), and 82 % of the acoustic energy of the chorus was within the range of 700 to 2000 Hz (Radford et al. 2008b). Thus, the assumption that crabs and fish (both larvae and adults) are capable of detecting the low frequencies emitted from vessels, offshore wind turbines and tidal turbines may be viable. The sound transmitted from the local causeway (400 – 800 Hz) is concluded to be of little impact to fishes and invertebrates around the area due the estimated intensity (127 dB re 1 µPa @ 1 m) being too low to mask background sound.
Table 2.4.1: A comparison of different vessel and turbine sounds and their acoustic characteristics.

<table>
<thead>
<tr>
<th>Anthropogenic sound source</th>
<th>Source level (dB re 1 µPa @ 1 m)</th>
<th>Peak frequency (Hz)</th>
<th>Band width (Hz)</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Small craft and boats</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Class 1 power boat</td>
<td>180</td>
<td>200</td>
<td>&lt; 5000</td>
<td>Amoser et al. 2004</td>
</tr>
<tr>
<td>Griffon 2000TD Hovercraft (12 m long, 40 km h⁻¹)</td>
<td>133</td>
<td>87</td>
<td>10-10000</td>
<td>Blackwell &amp; Greene 2005</td>
</tr>
<tr>
<td>Zodiac with outboard engine</td>
<td>162</td>
<td>-</td>
<td>20-20000</td>
<td>Erbe 2002</td>
</tr>
<tr>
<td>Arete Cabin Cruiser, (8 m long, 24 km h⁻³)</td>
<td>159</td>
<td>-</td>
<td>-</td>
<td>Erbe 2002</td>
</tr>
<tr>
<td></td>
<td>172</td>
<td>400, 2000</td>
<td>50-20000</td>
<td>300</td>
</tr>
<tr>
<td></td>
<td>152</td>
<td>100</td>
<td>10000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>150</td>
<td>300</td>
<td>250-1000</td>
<td>1000-5000</td>
</tr>
<tr>
<td></td>
<td>160</td>
<td></td>
<td>70-10000</td>
<td></td>
</tr>
<tr>
<td>Cabin cruiser, (8.5 m long, 11 km h⁻¹)</td>
<td>184</td>
<td>70, 100-800</td>
<td>100-2000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>188</td>
<td>5000</td>
<td>50-10000</td>
<td></td>
</tr>
<tr>
<td>Fishing vessel, (12 m long, 13 km h⁻¹)</td>
<td>191</td>
<td>80, 200, 500</td>
<td>80-2000</td>
<td></td>
</tr>
<tr>
<td>Small boat outboard engine</td>
<td>185</td>
<td>100, 1000</td>
<td>80-1200</td>
<td></td>
</tr>
<tr>
<td><strong>Medium sized ships</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spirit of Alaska tour vessel, (44 m long, 19 km h⁻¹)</td>
<td>180</td>
<td>200, 5000</td>
<td>70-20000</td>
<td>Kipple &amp; Gabriele 2004</td>
</tr>
<tr>
<td>Nunatak, Naval vessel, (20 m long, 26 km h⁻¹)</td>
<td>169</td>
<td>400-630</td>
<td>50-20000</td>
<td>Kipple &amp; Gabriele 2003</td>
</tr>
<tr>
<td>HMNZS Wellington Naval vessel, (85 m long, 15km h⁻¹)</td>
<td>172</td>
<td>50, 400</td>
<td>50-5000</td>
<td></td>
</tr>
<tr>
<td><strong>Large sized ships</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Golden Bay container vessel, (98 m long, 24 km h⁻¹)</td>
<td>192</td>
<td>100-400</td>
<td>100-800</td>
<td></td>
</tr>
<tr>
<td>Maersk Batur container vessel, (290 m long, 15 km h⁻¹)</td>
<td>186</td>
<td>100-400</td>
<td>100-1200</td>
<td></td>
</tr>
<tr>
<td>Buxlink container vessel, (206 m long, 11 km h⁻¹)</td>
<td>159</td>
<td>800-2000</td>
<td>100-2000</td>
<td></td>
</tr>
<tr>
<td>Container vessel, (173 m long, 29 km h⁻³)</td>
<td>192</td>
<td>-</td>
<td>40-100</td>
<td>Hildebrand 2009</td>
</tr>
<tr>
<td>Container vessel, (270 m long)</td>
<td>198</td>
<td>23</td>
<td>5-100</td>
<td>Hildebrand 2004</td>
</tr>
<tr>
<td>Coal carrier vessel, (173 m long, 29 km h⁻³)</td>
<td>192</td>
<td>50-100</td>
<td>-</td>
<td>Arveson &amp; Vendittis 2000</td>
</tr>
<tr>
<td><strong>Turbines</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Utgrunden offshore wind turbine, (13 m s⁻¹ wind speed)</td>
<td>151</td>
<td>180</td>
<td>&lt; 1000</td>
<td>Walhberg &amp; Westerberg 2005</td>
</tr>
<tr>
<td>Middelgrunden offshore wind turbine (6 m s⁻¹ wind speed)</td>
<td>109</td>
<td>25</td>
<td>12.5-500</td>
<td>Tougaard &amp; Miller 2009</td>
</tr>
<tr>
<td>Bockstigen-Valar offshore wind turbine (8 m s⁻¹ wind speed)</td>
<td>113</td>
<td>160</td>
<td>12.5-500</td>
<td>Tougaard &amp; Miller 2009</td>
</tr>
<tr>
<td>Vindeby offshore wind turbine (13 m s⁻¹ wind speed)</td>
<td>127</td>
<td>25</td>
<td>12.5-500</td>
<td>Tougaard &amp; Mille 2009</td>
</tr>
<tr>
<td>Lynmouth Underwater tidal turbine</td>
<td>175</td>
<td>5000</td>
<td>8000</td>
<td>Parvin et al. 2005</td>
</tr>
</tbody>
</table>
The distance at which a given animal may hear the sound from a vessel or turbine is very difficult to answer. This is because detection distances for vessels, offshore wind turbines and tidal turbines depend on the water depth, substrate type, weather conditions, the size of the turbine or vessel, the velocity of the vessel, as well as the animal’s hearing range and thresholds (Wahlberg & Westerberg 2005). Furthermore, the parameters required to calculate such detection distances are poorly understood for fishes (Wahlberg & Westerberg 2005) and even more so for crustaceans. The estimated detection distance for all vessels in this study varied from 783 m (the *HMNZS Wellington*) to 1.4 km (the *Golden Bay* container vessel) in the inner Gulf, and 3.2 km to more than 31 km in the outer Gulf. The sound from the tidal turbine was estimated to be detectable within 2 km from the turbine and therefore can potentially mask acoustic signals and ambient sound like vessels. Wind turbines are likely to have a more localised impact as source levels are far smaller (151 dB re 1 µPa @ 1 m) (Walhberg & Westerberg 2005) and have only small detection distances below 100 m (based on the calculations in this study). The detectable distance at which fish can detect the sound from offshore wind turbines is highly dependent on the hearing range of the fish species and thresholds. It has been estimated that the goldfish *Carassius auratus*, the cod *Gadus morhua* and the Atlantic salmon *Salmo salar* can detect low-frequency wind turbine sound (operating during 8 m s⁻¹ wind speed) (63 Hz – 100 Hz) at a distance of 25, 13, and 0.4 km respectively (Wahlberg & Westerberg 2005). In higher wind speed conditions (13 m s⁻¹), the same turbine can be detected by the same species at a distance of 15 km (*C. auratus*), 7 km (*G. morhua*), and 0.5 km (*S. salar*). The sound from turbines has also been found incapable of masking the communication within harbour seals and porpoises (Tougaard et al. 2009). Physiological damage, such as hearing loss, was found to be improbable for fishes, as within 10 m from the pylon structure, the intensity is below what is required for hearing damage in fish (Walhberg & Westerberg 2005). Based on the *TL* model presented in this study, at 10 m distance from the pylon structure the sound intensity level from an offshore wind turbine is only 118 dB re 1 µPa. However, the conclusions reported by Walhberg & Westerberg (2005) are based on data which encompass large uncertainties.

The installation of tidal turbines in the Kaipara Harbour will be a significant source of anthropogenic sound. Spectral composition for an operational tidal turbine shows greatest intensity between 800 Hz and 8 kHz, at a high intensity. Over the next five years, 200 tidal turbines are scheduled to be installed in the Kaipara Harbour. Given that the turbine sound
can travel approximately 2 km, it is probable that the sound from several turbines will overlap. Consequently, the intensity and spectral composition of turbine sound in the field may be different than presented in the current study due to additive and cancellation effects of sound from neighbouring turbines. An environmental impact assessment report for the Kaipara Harbour by Crest Energy Ltd (2006) estimates detection distances for a MW tidal turbine of several hundred metres, but crudely assumes only spherical spreading which is unlikely in the relatively shallow water where surface and seafloor grazing angles are more likely to result in sound propagating somewhere between spherical and cylindrical spreading with peaks and troughs with distance. Furthermore, attenuation coefficients and other parameters are excluded in the estimations. In the current study, a detection distance of 2 km is more accurate, however, it is the subject of many limitations and should be interpreted carefully for a number of reasons. Firstly, the hearing thresholds and frequencies are unknown for crustaceans and many fishes which inhabit the Kaipara. Secondly, the propagation of sound will vary in time and space due to weather, seasons, tidal currents, depths and bottom substrates. Thirdly, ambient sound levels also vary in time and space due to dawn and evening choruses, seasonality and local fauna. Fourthly, the strength of the tidal current will alter SL’s of the tidal turbine. Also, the type of tidal turbine will also alter the sound output and composition of sound (Parvin et al. 2005). For example, turbines equipped with gearboxes reportedly produce greater source levels compared to turbines fitted with hydraulic mechanisms (Parvin et al. 2005). Consequently, the detectable distance at which a given crustacean or fish species may hear the turbine may be under or overestimated in this study depending on the environmental conditions at the time. These same assumptions also apply to vessel sound. Vessel specifications and velocity are not the only reasons why detection distances may be under- or overestimated. Ambient levels will increase during the summer months as the number of operating boats on the Hauraki Gulf will rise. Also, within the summer months, there will be significant diurnal variation. This would not only be due to elevated levels of boating activity during the summer months, but also increased biological activity on reefs as spawning season approaches for many fishes and invertebrates. The sound from the motorway causeway also varies depending on traffic flow, time of day (peak hour traffic) and weather. As vessel and turbine sound propagates through the water, the spectral composition will also change. High frequencies will attenuate out and the sound will become a low hum, dominated by frequencies below 1 kHz (Richardson et al. 1995; Au & Hastings 2008). Alternatively, the shallow water may prevent low frequencies from successfully
propagating and sea surface and bottom substrate will heavily influence the propagation of any particular anthropogenic sound (Au & Hastings 2008).

Temporary hearing loss (referred to as a temporary threshold shift, or TTS (Popper & Hastings 2009)) in crustaceans have not been studied, so it is difficult to say whether the sound from the Superflyte, Kawai Kat, R.V. Yellow Naiad, R.V. Hawere, HMNZS Wellington, and the container vessels would have such effects. However, high intensity sounds are able to fatigue, damage or kill the sensory hair cells within the auditory organs in fish producing temporary or permanent hearing damage (Popper & Hastings 2009). For example, the sound emitted from an 8.5 m, 163 hp inboard engine cabin cruiser (SL of 152 dB re 1 µPa @ 1 m) has been found to substantially reduce auditory sensitivity relative to the hearing thresholds in ambient sound in Chromis chromis, Sciaena umbra, and Gobius cruentatus (Codarin et al. 2009). A temporary threshold shift can occur when the receiver is subjected to either low levels of sound for a relatively long period of time, or a high level of sound for a shorter period of time (Popper & Hastings 2009). For example, the goldfish, C. auratus, had a 5 dB TTS after only 10 minutes of exposure to band limited sound (0.1 – 10 kHz, approximately 170 dB re 1 µPa overall SPL) (Smith et al. 2004a, b). Following three weeks of exposure, the same stimulus fish showed a 25 dB threshold shift and took more than two weeks to return to normal hearing. In another study, the fathead minnow, Pimephales promelas, was found not to return to normal hearing thresholds after 14 days following 24 hours of exposure to white sound (0.3 – 2.0 kHz, 142 dB re 1 µPa) (Scholik & Yan 2001).

The sound emitted from ferries and commercial vessels are unlikely to cause permanent hearing loss because the receiver is unlikely to be exposed to the high intensity artificial sound for periods greater than several minutes. Tidal turbines, however, emit high intensity sound during the tides, and, based on Smith et al. (2004) and Scholik & Yan (2001)’s findings, it may be possible for fishes to experience temporary hearing losses if close enough to the tidal turbine. However, there is no data to suggest such turbines may cause temporary or permanent physiological impairment, due to the lack of research. Similarly, due to the low attenuation of sound emitted from container vessels, it may be plausible to assume temporary physiological impairment to fishes, albeit such a statement goes without experimental evidence.
Vessel sound is one of the primary sources of sound pollution in the Hauraki Gulf and New Zealand. In future years, tidal and wind turbines may also contribute to sound pollution in certain areas. This study only examined the acoustic characteristics of several common types of coastal vessels, one proposed MW tidal turbine which is to be installed in the Kaipara Harbour, and a Danish offshore wind turbine. There was not much investigation into how these sounds may already be increasing ambient sound levels within the Hauraki Gulf or Kaipara Harbour. The realisation that these anthropogenic sounds can significantly raise background sound levels in harbours and the ocean is an important one that requires attention. While acoustic masking is one of the primary concerns of sound pollution, physiological stress and impairment is also causing concerns about how humans are changing the underwater environment.

Anthropogenic underwater sound in the Hauraki Gulf is of low frequencies and high source levels and can therefore propagate vast distances (albeit very dependent on vessel specifications, environmental factors and the audiology of the receiver). Offshore wind turbines are of low frequencies also, while tidal turbines produce a broader bandwidth sound that can travel several kilometres. All anthropogenic sound measured in this study, except motorway causeways, in the Hauraki Gulf (and other areas) have potential in masking biologically important sound that help many post-larval crab and fish species orient towards a suitable settlement habitat (Radford et al. 2007) and settle (Stanley et al. 2010). Also, tidal turbine sound and container vessels have the potential to cause temporary hearing loss in fishes and physiological stress to a wide variety of fishes and crustaceans that have hearing thresholds within 6 dB above background sound levels for 0.1 – 5 kHz and audible frequency ranges below 5 kHz.
Chapter Three

The influence of anthropogenic sound on the settlement behaviour in estuarine crab megalopae

Published as:

3.1 Introduction

Underwater tidal turbine technology has advanced at a rapid rate due to increasing commercial interest across many countries. This is the result of a widely recognised need to shift energy production from fossil fuels to renewable sources in order to limit further anthropogenically induced climate change (King 2004; Rosenzweig et al. 2008; Inger et al. 2009). Tidal power generation is an emerging renewable energy technology, and many wind turbines are already in place within coastal waters of numerous countries (Herbert et al., 2007) and a few pilot projects on underwater tidal turbines (Ferro 2006; Cada et al. 2007; Inger et al. 2009).

The advantages of renewable energy generation are not in doubt; however, locally the environmental impacts can be significant and need to be carefully considered (Inger et al. 2009). While wind turbine farms in coastal waters are well established in Northern Europe and their environmental impacts have been studied to some extent, underwater tidal turbines are still in their infancy and their impacts are largely unknown (Inger et al. 2009). The impact of anthropogenic underwater sound on marine life is of growing concern, with an increasing body of evidence indicating negative impacts (Inger et al. 2009; Thomas 2009; Slabbekoorn et al. 2010). The sound generated during the construction and installation of turbine farms has already been identified as being of concern as pile-driving has been observed to directly
impact cetaceans and fishes (Inger et al. 2009; Thomas 2009). Very little is understood about the operational sound of underwater tidal turbines and further research is required before drawing conclusions on how their sound will influence marine life (Inger et al. 2009). The underwater sound from tidal turbines will be influenced by several factors, including blade and turbine design, tidal flow velocity, depths, bottom substrate, gearboxes, and weather (Lloyd et al. 2011). Similarly, wind speeds and turbine technology also influences the sound generated from operating offshore wind turbines (Wahlberg & Westerberg 2005). Therefore, the sound generated and its impacts will be specific to sites and generating devices (Lloyd et al. 2011). The sound from an operating ‘SeaFlow’ tidal turbine has been measured to have a source level of approximately 175 dB re 1 µPa @ 1 m with peak intensities at 0.1, 0.8, 2, 5 and 8 kHz (Parvin et al. 2005) at a maximum tidal flow of approximately 3 m s\(^{-1}\) (Richards et al. 2007; Lloyd et al. 2011). Offshore wind turbines have been found to have an operational underwater source level of 154 dB re 1 µPa @ 1 m at a wind speed of 13 m s\(^{-1}\) (Wahlberg & Westerberg 2005). These source levels are louder than the ambient underwater sound levels commonly encountered in coastal waters. Consequently, the addition of these anthropogenic sound sources is likely to result in the masking of underwater ambient sound for organisms that rely on acoustic communication or natural acoustic cues within this frequency range (Slabbekoorn et al. 2010).

The life history of brachyuran crabs typically involves a planktonic larval stage that ends with a post-larva, or megalopa, that actively swims to find suitable benthic habitat in which to settle and develop into a reptant juvenile (Simpson et al. 2005; Mann et al. 2007; Radford et al. 2007; Radford et al. 2008; Radford et al. 2010; Stanley et al. 2010; Simpson et al. 2011). To help ensure megalopae settle in a suitable location, they have evolved the ability to detect and orient toward physical and chemical cues associated with their preferred benthic habitats (Pawlik 1992; Steinberg et al. 2008; Medina & Tankersley 2010). Once megalopae encounter their preferred habitat, settlement and subsequent metamorphosis from the megalopa to juvenile is often instigated by a combination of several physical and chemical cues, which can include acoustic cues (Forward et al. 2001; Gebauer et al. 2004; Steinberg et al. 2008; Stanley et al. 2010).

The duration of the megalopal stage can be relatively plastic and may depend on the presence or absence of several settlement cues (Pechenik 1990; Gebauer et al. 1999; Stanley et al. 2010).
For example, in the presence of estuarine water the megalopae of the blue crab, *Callinectes sapidus*, decrease their time to metamorphosis (TTM), i.e., the time taken for the larva to moult from a megalopa to the first instar juvenile crab (Forward et al. 1996). However, delaying metamorphosis for too long (beyond a specific temporal threshold) may result in the death of the megalopa, or result in spontaneous metamorphosis of the larva followed by poor subsequent post-settlement growth (Pechenik 1990; Weber & Epifanio 1996; Gebauer et al. 1999; Forward et al. 2001; Steinberg et al. 2008; Stanley et al. 2010). These temporal thresholds are typically determined in the laboratory by rearing megalopae in control treatments of “untainted” seawater (Stanley et al; 2010). Depending on the species, the TTM of megalopae can typically be shortened by approximately 15 to 60% upon exposure to appropriate settlement cues (Forward et al. 2001; Stanley et al. 2010). For example, when subjected to reef sound the megalopae of five common coastal species of reef-dwelling brachyuran crabs all accelerated their physiological development and TTM was reduced by between 34 and 60% (Stanley et al. 2010). These results suggest that natural underwater sound plays an important role in the metamorphosis of brachyuran crabs, especially coastal reef dwelling species. However, no data have been published to suggest the same responses are seen in estuarine species when exposed to estuarine sound. The characteristics of underwater sound that are responsible for expediting metamorphosis in crabs are unknown at this time, but may involve sound intensity, frequency composition, or temporal variability in both frequency and intensity, or any combination of these acoustic characteristics. Furthermore, it is possible that other sources of underwater sound may elicit or interfere with the normal metamorphosis response of megalopae in relation to natural acoustic cues. The biological effects of anthropogenic sound in the underwater environment have become of increasing interest in response to rising levels of anthropogenic sound in coastal and ocean waters (Thomas 2009; Slabbekoorn et al. 2010).

The effects of anthropogenic sound on marine mammals and adult fishes have been well studied (Thomas 2009; Slabbekoorn et al. 2010). However, very few studies have dealt with larvae of marine organisms, and none have investigated the effect of anthropogenic sound on the settlement and metamorphosis of crustacean larvae. Furthermore, no experimental data have been published which investigate the metamorphosis response of estuarine crab megalopae to ambient mudflat sound or the possible effect of tidal and wind turbine sound on their metamorphosis behaviour.
Therefore, the aim of the current research was three-fold: (1) to determine the metamorphosis response of the megalopae of two common estuarine crabs in New Zealand, *Austrohelice crassa* and *Hemigrapsus crenulatus*, to natural ambient estuarine sound; (2) to determine whether the underwater sound emitted from tidal and wind turbines influences the metamorphosis response of the crab megalopae; and (3) to attempt to identify which characteristics of turbine sound are responsible for eliciting any observed changes in metamorphosis behaviour of the megalopae.
3.2 Methods

3.2.1 Sourcing crab megalopae for behavioural assays

All experiments were completed between May 2011 and May 2012 using light traps to capture pelagic crab megalopae (Stanley et al. 2010). Up to three traps were deployed at night at the same location along the coast at Leigh, in north-eastern New Zealand. The following morning the captured megalopae were placed in containers filled with saltwater and transported back to the laboratory where they were identified, counted and sorted by settlement stage. Only intermoult pre-settlement (i.e., natant and actively swimming) megalopae were selected for use in each experimental assay. If a trap contained large planktivorous fishes, megalopae were not used as they may have altered behaviour due to being in the presence of a predator (Forward & Rittschof 2000; Stanley et al. 2010). Selected megalopae were contained in UV treated and filtered (40 µm) seawater in the laboratory, under natural photoperiod and ambient temperature until experiments begun at 17:00 hrs on the day of capture (Stanley et al. 2010).

3.2.2 Behavioural assays

Each experimental assay consisted of silent (control, as no settlement cues) and sound treatments, all of which were contained within the same laboratory, but acoustically isolated by the use of foam rubber mats. The absence of any acoustic signals transmitting from the sound treatments to the silent treatments was confirmed by recording with a calibrated omnidirectional hydrophone (HTI-96 min, High Tech Inc., USA).

Experimental replicates consisted of a water bath (used to maintain constant water temperature) which was used to hold up to ten 250 mL plastic vials with sealed lids that each contained an individual megalopa in 230 mL of 1 µm filtered and UV treated seawater (Figure 3.2.1). Any one treatment consisted of three replicates and each replicate contained at least seven megalopae, i.e., at least 21 megalopae per treatment. The number of treatments per experiment varied between two and four depending on the availability of megalopae from the light traps.
Figure 3.2.1: Experimental set up of treatments: (A) Schematic diagram of the settlement assay. Large boxes represent the water baths and numbered circles represent each 250 mL vial holding an individual megalopa. (B) Cross-sectional diagram of one treatment consisting of the water bath, underwater loudspeaker and housing vials for megalopae.
Megalopae require roughened substrates to settle (Stanley et al. 2010) and thus each vial had a roughened base to simulate a chemically inert settlement substrate for megalopa. Both the silent and sound treatments contained a Phillips loudspeaker inside a water tight plastic bag on the bottom of the water bath (Stanley et al. 2010). For the sound treatments, an MP3 player was connected to a Phillips SBA1500 amplifier and speaker to continuously playback a 10 min loop of recorded turbine or mudflat sound into the water bath which was also transmitted through the acoustically transparent plastic vials holding the megalopae.

When sufficient megalopae (at least 21) of the same species and similar settlement stage were captured in the light traps, an individual larva was placed in each 250 mL plastic vial and the vials were then randomly allocated to the water baths. Sound and silent treatments were also randomly allocated to water baths for each experiment.

Once all megalopae were transferred to experimental treatments, the MP3 player was turned on to begin playback in a sound treatment – signifying the beginning of the experiment. Every six hours, the individual crabs were examined to determine if they had settled and metamorphosed into a first instar juvenile crab. The period from the commencement of the experiment until the appearance of the first instar juvenile in each vial was termed ‘time to metamorphosis’ (‘TTM’ (Stanley et al. 2010)). When counts were made during the night, a dark red light was used to minimise disturbance of the crabs (Jeffs & Holland 2000).

The experiment was terminated when all megalopae in all replicates for all treatments had settled and metamorphosed into first instar juvenile crabs. No mortality events occurred.

3.2.3 Underwater sound recordings for playback experiments

A recording of a tidal turbine was not possible to obtain because there are only a few operational tidal turbines anywhere in the world and operators with recordings of turbines refused to supply them for this study. Thus a digital analogue, which matched the same frequency composition and peak intensities, was used for the sound treatments and was based on a published spectrum of a tidal turbine operating under a maximum tidal flow of 3 ms-1 (Parvin et al. 2005).
Underwater recordings from the Utgrunden coastal wind farm in Denmark were used during playback experiments and provided by Dr. Jakob Tougaard from the National Environmental Research Institute, Denmark.

Ambient underwater sound was recorded in February 2012 during late evening (19:00 – 21:00 hrs) chorus within a subtidal mudflat habitat in the southern arm of Kaipara Harbour where both experimental crab species are found in abundance, including large numbers of juveniles (S 36° 24′ 36.5″ E 174° 22′ 40.9″). Calibrated High Tech, Inc. HTI-96 omnidirectional hydrophones (10 Hz to 60 kHz flat response) connected to a watertight temporal recording unit (20 dB gain, 16 bit, 48 kHz sampling rate) were used to record mudflat sound.

Before each experiment began, a calibrated hydrophone (HTI-96 min, High Tech Inc. USA) was used to adjust the source level produced from the Phillips loudspeakers in each replicate sound treatment to the desired sound level (either 145 or 125 dB re 1 µPa for turbine treatments or 125 dB re 1 µPa for mudflat treatment). These levels were used because 145 dB re 1 µPa was the greatest output level achievable with the speaker and was as close as possible to the published source levels of an operating tidal (175 dB re 1 µPa (Parvin et al. 2005) and wind turbine (154 dB re 1 µPa (Wahlberg & Westerberg 2005)). An output level of 125 dB re 1 µPa for the mudflat treatment was selected as this was the measured mean (over 2 hours) ambient sound level for that habitat during dusk in summer. An output level of 125 dB re 1 µPa for a tidal turbine sound treatment was also used in one experiment to match the intensity level of mudflat sound treatment to determine if sound level alone was responsible for influencing metamorphosis behaviour in crab megalopae (refer to table 1).

Unfortunately, comparisons of TTMIs between experiments were not appropriate due to an inability to accurately determine the starting ages of the wild-caught megalopae. As such, a series of seven experimental combinations were necessary because of the vagaries of supply of wild megalopae. The seven experiments each tested an individual combination of experimental treatments. Comparisons among treatments were possible within individual experiments as all subject megalopae were from the same wild-caught cohort and were randomly assigned to experimental treatments and replicates.
3.2.4 Data analyses

Nonparametric statistical methods were used to analyse the differences between median TTM values within and among treatments. Mann-Whitney tests or a Kruskal-Wallis one-way analysis of variance on ranks were used to test for differences in the median TTM among replicates within individual treatments (i.e., a separate analysis for each treatment). If these comparisons were not significant for each treatment, then the TTMs for all replicate tanks within each treatment were pooled and used to compare the median TTMs among the treatments (Stanley et al. 2010). For all statistical comparisons, a $P$ value $\leq 0.05$ was considered significant. Dunn’s pairwise multiple comparisons tests were used to determine differences in the median TTMs between individual pairs of treatments where the overall experiment had been found to contain significant differences among treatments. All statistical analyses were carried out using the statistical software Sigma Plot 11.0 and Minitab 15.0.
3.3 Results

3.3.1 Confirmation of sound sources

For the wind or tidal turbine sound exposure treatments, the resulting sound in the experimental tanks was of an overall similar spectral composition to the source signals (Figure 3.3.1). Broadcasted mudflat sound replayed into replicate experimental tanks also matched the overall spectral composition and intensity of the *in situ* recording (Figure 3.3.1). Hydrophone recordings from the silent controls confirmed the absence of any sound being transmitted from sound treatments or external sources (Figure 3.3.1).

3.3.2 Pooling of replicates

There was no significant difference in the median TTM among individual replicates within both the sound and the silent treatments for all seven metamorphosis experiments (Kruskal-Wallis test) (Table 3.3.1). Therefore, in all experiments the results from individual replicates for each treatment were able to be pooled together for comparison between the pooled results from other treatments within each experiment.

3.3.3 Effect of mudflat sound on the TTM

The megalopae of both crab species showed a significantly shorter median TTM when exposed to mudflat sound compared to the silent control with *A. crassa* and *H. crenulatus* showing a 31 % ($H_3 = 29.13, P < 0.001$) (Table 3.3.2, experiment 1) and 21 % ($H_3 = 23.23, P < 0.001$) (Table 3.3.2, experiment 2) reduction in median TTM, respectively.

3.3.4 Effect of turbine sounds on the TTM

Both wind and tidal turbine sound at levels of 145 dB re 1 µPa caused a significantly longer median TTM in the megalopae of *A. crassa*, and *H. crenulatus*, compared to silent control treatments (Table 3.3.2, experiment 3, 4, 5, & 6).
The megalopae of *A. crassa* that were subjected to tidal turbine sound at 145 dB re 1 µPa showed an increase in TTM of approximately 26 %, compared to the silent control treatment (Mann-Whitney *U* test, *P* = 0.006) (Table 3.3.2, experiment 3). The megalopae of *H. crenulatus* also showed a significant increase (19 %) in TTM when subjected to tidal turbine sound compared to the silent control (Mann-Whitney *U* test, *P* = 0.042) (Table 3.3.2, experiment 4).

Compared to silent control treatments, wind turbine sound at a level of 145 dB re 1 µPa was also found to delay metamorphosis in both *A. crassa* and *H. crenulatus*, with an increase in median TTM by 15 % (Mann-Whitney *U* test, *P* = 0.006) (Table 3.3.2, experiment 5) and 24 % (Mann-Whitney *U* test, *P* = 0.042) (Table 3.3.2, experiment 6), respectively.

### 3.3.5 Effect of mudflat sound versus anthropogenic sound on TTM

When *A. crassa* megalopae were exposed to mudflat sound at the same level as *in situ* mudflat sound (i.e., 125 dB re 1 µPa), the median TTM decreased by 47 % when compared to the tidal turbine sound treatment, and 46 % compared to wind turbine treatments (*H* = 29.13, *P* < 0.001) (Table 3.3.2, experiment 1). Similarly, *H. crenulatus* megalopae showed decreases of 38 % and 40 % when exposed to mudflat sound compared to tidal and wind turbine sound, respectively (*H* = 23.23, *P* < 0.001) (Table 3.3.2, experiment 2).

### 3.3.6 Effect of turbine sound intensity on median TTM

There was no significant difference in median TTM between *A. crassa* megalopae exposed to tidal turbine sound at a source level of either 145 or 125 dB re 1 µPa (Mann-Whitney *U* test, *P* = 0.69) (Table 3.3.2, experiment 7; Figure 3.3.2). However, the median TTM in *A. crassa* megalopae in both sound level treatments (i.e., 145 and 125 dB re 1 µPa) were significantly longer than the silent control by 17 – 22 % (Mann-Whitney *U* test, *P* < 0.05) (Table 3.3.2, experiment 7).
Figure 3.3.1: Spectral composition of experimental sound sources and when played back in experimental tanks: A) Sound recorded in silent treatment tanks (control); B) Mudflat recording from the Kaipara Harbour, New Zealand; C) Underwater tidal turbine; D) Wind turbine in coastal waters of Denmark. Black lines represent digital analogue (as in the case for tidal turbine sound) or in situ recordings (as in the case of wind turbine and mudflat sound).
### Table 3.3.1: Summary from seven individual experiments of comparisons of median TTM values among replicates within each treatment. Kruskal-Wallis test showing no significant difference for replicates within all experimental treatments ($P > 0.05$).

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Species</th>
<th>Sample size (n)</th>
<th>Treatment</th>
<th>$P$-value</th>
<th>H-statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Austrohelice crassa</td>
<td>30</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>0.56</td>
<td>1.08</td>
</tr>
<tr>
<td></td>
<td></td>
<td>27</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>1.00</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td></td>
<td>30</td>
<td>Mudflat (125 dB re 1 µPa)</td>
<td>0.48</td>
<td>1.45</td>
</tr>
<tr>
<td></td>
<td></td>
<td>30</td>
<td>Silent</td>
<td>0.19</td>
<td>3.31</td>
</tr>
<tr>
<td>2</td>
<td>Hemigrapsus crenulatus</td>
<td>21</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>0.43</td>
<td>1.67</td>
</tr>
<tr>
<td></td>
<td></td>
<td>21</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>1.00</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td></td>
<td>21</td>
<td>Mudflat (125 dB re 1 µPa)</td>
<td>0.07</td>
<td>5.43</td>
</tr>
<tr>
<td></td>
<td></td>
<td>21</td>
<td>Silent</td>
<td>0.81</td>
<td>0.41</td>
</tr>
<tr>
<td>3</td>
<td>Austrohelice crassa</td>
<td>30</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>0.58</td>
<td>1.08</td>
</tr>
<tr>
<td></td>
<td></td>
<td>30</td>
<td>Silent</td>
<td>0.33</td>
<td>2.44</td>
</tr>
<tr>
<td>4</td>
<td>Austrohelice crassa</td>
<td>27</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>0.10</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td></td>
<td>27</td>
<td>Silent</td>
<td>0.59</td>
<td>1.07</td>
</tr>
<tr>
<td>5</td>
<td>Hemigrapsus crenulatus</td>
<td>30</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>0.43</td>
<td>1.67</td>
</tr>
<tr>
<td></td>
<td></td>
<td>30</td>
<td>Silent</td>
<td>0.91</td>
<td>0.20</td>
</tr>
<tr>
<td>6</td>
<td>Hemigrapsus crenulatus</td>
<td>21</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>1.00</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td></td>
<td>21</td>
<td>Silent</td>
<td>0.81</td>
<td>0.41</td>
</tr>
<tr>
<td>7</td>
<td>Austrohelice crassa</td>
<td>27</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>0.79</td>
<td>0.46</td>
</tr>
<tr>
<td></td>
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<td>0.26</td>
<td>2.68</td>
</tr>
<tr>
<td></td>
<td></td>
<td>27</td>
<td>Silent</td>
<td>0.21</td>
<td>3.15</td>
</tr>
</tbody>
</table>
Table 3.3.2: Comparisons among median TTM for each treatment for two estuarine crab species from seven individual sound exposure experiments. Different superscript letters indicate significant difference between median TTMs within an individual experiment ($P < 0.05$).

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Species</th>
<th>Treatment</th>
<th>Median TTM (h)</th>
<th>Difference from Silent control TTM (h)</th>
<th>$P$-value</th>
<th>$H$-statistic*</th>
<th>$U$-value**</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Austrohelice crassa</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>114</td>
<td>27&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>111</td>
<td>24&lt;sup&gt;b&lt;/sup&gt;</td>
<td>&lt; 0.001</td>
<td>29.129*</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mudflat habitat (125 dB re 1 µPa)</td>
<td>60</td>
<td>27&lt;sup&gt;c&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>87</td>
<td>0&lt;sup&gt;c&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Hemigrapsus crenulatus</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>144</td>
<td>54&lt;sup&gt;b&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>150</td>
<td>60&lt;sup&gt;b&lt;/sup&gt;</td>
<td>&lt; 0.001</td>
<td>23.229*</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mudflat habitat (125 dB re 1 µPa)</td>
<td>90</td>
<td>24&lt;sup&gt;c&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>114</td>
<td>0&lt;sup&gt;c&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Austrohelice crassa</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>114</td>
<td>30&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.006</td>
<td>234.0**</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>84</td>
<td>0&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Austrohelice crassa</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>156</td>
<td>24&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.04</td>
<td>238.5**</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>132</td>
<td>0&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Hemigrapsus crenulatus</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>126</td>
<td>24&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.006</td>
<td>189.5**</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>102</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>Hemigrapsus crenulatus</td>
<td>Wind turbine (145 dB re 1 µPa)</td>
<td>150</td>
<td>36&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.04</td>
<td>141.0**</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>114</td>
<td>0&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>Austrohelice crassa</td>
<td>Tidal turbine (145 dB re 1 µPa)</td>
<td>132</td>
<td>24&lt;sup&gt;b&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tidal turbine (125 dB re 1 µPa)</td>
<td>126</td>
<td>18&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.025</td>
<td>7.348*</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silent control</td>
<td>108</td>
<td>0&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 3.3.2: Percentage (%) of total megalopae to metamorphose against time (hours). *Austrohelice crassa* experiments: (A) experiment 3; (B) experiment 4; (C) experiment 1; (D) experiment 7. *Hemigrapsus crenulatus* experiments: (E) experiment 5; (F) experiment 6; (G) experiment 2.
3.4 Discussion

International interest in renewable energy production using tidal and wind turbines is growing extremely rapidly. However, there has been limited research into the environmental impact of these technologies, especially the impact of emitted underwater sound on marine life. Natural sources of underwater sound have previously been found to play an important role in influencing the settlement of many coastal organisms, including the megalopae of many coastal crab species (Radford et al. 2007; Stanley et al. 2010), as well as fish, mussel and coral larvae (Simpson et al. 2005; Vermeij et al. 2010; Wilkens et al. 2012). Therefore, there is the potential for the underwater sound from wind and tidal turbines installed in shallow water habitats, to interfere with these natural acoustic settlement cues. The present study found natural mudflat sound to consistently reduce the median TTM compared to silent controls by 21 – 31 % in two crab species A. crassa and H. crenulatus which are common inhabitants of soft-shore habitats in New Zealand. In comparison, when A. crassa megalopae were previously experimentally exposed to underwater reef sound they showed no significant reduction in TTM compared to the silent control, which suggests that this species has habitat-specific sound cues for settlement, as have been found in other coastal brachyuran crab species (Stanley et al. 2011).

Underwater sound from turbines with a source level of 145 dB re 1 µPa was found to delay metamorphosis of the megalopae of both crab species by 27 – 31 % for tidal turbine sound and 27 – 32% for wind turbine sound, compared to silent control treatments. A delay in metamorphosis may prevent megalopae from settling into suitable habitats and will result in them spending more time in the plankton which is likely to increase their already high risk of predation (O'Connor 1991; O'Connor & Gregg 1998; Stanley et al. 2010). This could lead to lower recruitment of crab species within estuaries and other soft-shore habitats in the vicinity of coastal turbines. Delayed metamorphosis due to underwater sound from turbines may also be an issue in any other species which have sensitivity to acoustic settlement cues, such as coral, mussels and fish (Simpson et al. 2005; Montgomery et al. 2006; Vermeij et al. 2010; Wilkens et al. 2012). Furthermore, the interference in the metamorphosis responses in crab megalopae when subjected to varying intensity levels of tidal turbine sound may also suggest that other continuous anthropogenic underwater sound sources of similar frequency
composition and intensity, such as shipping (most acoustic energy below 1 kHz (Wright 2008), may have similar effects on settlement and metamorphosis in crustacean megalopae.

While these results suggest turbine sound may mask natural acoustic settlement cues, the spatial scale over which such masking may occur is difficult to infer because little is known about acoustic detection thresholds of crustaceans. Previous research has investigated the acoustic settlement response thresholds in the megalopae of a range of brachyuran crab species and these were found to vary substantially among species (Stanley et al. 2011). For example, the megalopae of *Leptograpsus variegatus*, *Cyclograpsus lavauxi* and *Hemigrapsus sexdentatus* showed behavioural response thresholds of 90, 100 and 126 dB re 1 µPa, respectively, to acoustic settlement cues from preferred settlement habitat (Stanley et al. 2011). Given these measured behavioural thresholds, the associated distances these crab species may be able to detect and respond to acoustic settlement cues were estimated at 199 and 39,811 m assuming spherical and cylindrical spreading of sound from the source, respectively (Stanley et al. 2011). Acoustic behavioural response thresholds for *A. crassa* or *H. crenulatus* are not known, however, if response thresholds are assumed to be similar to *L. variegatus*, *C. lavauxi* and *H. sexdentatus*, and the same cylindrical spreading and transmission loss models from past studies are applied, then the potential impact of turbine sound delaying metamorphosis could range for up to 40 km from the turbine source.

Besides acting as a settlement cue, natural sources of underwater sound from suitable settlement habitats also have a strong influence on the swimming behaviour in crab megalopae, with crabs orienting their swimming toward the sound source, presumably to assist in locating suitable settlement habitats (Radford et al. 2007). Although not examined in this study, it seems likely that underwater turbine sound may also interfere with the orientation behaviour of swimming crab megalopae, in the same manner it has been shown to interfere with their acoustic metamorphosis response. Testing this possibility warrants further research as it has the potential to have a greater influence of the spatial distribution of settling crab larvae in relation to underwater turbines.

Poorer recruitment and subsequently smaller local populations of estuarine crabs may have ecological effects due to their extremely high abundances (i.e., over 550 m$^{-2}$ for *A. crassa* (Jones & Simons 1983; Morrisey et al. 1999)), importance in bioturbation and nutrient
cycling in shallow waters (Sivaguru 2000), and as a food source for many commercially important coastal fishes (Gunson 1993).

Metamorphosis in both *A. crassa* and *H. crenulatus* appeared to be delayed beyond the assumed temporal threshold (theoretically represented by control treatments (Stanley et al. 2010) by at least 18 h when subjected to turbine sounds. This may be due to metamorphosis being delayed due to perceived unfavourable conditions, or because of an absence of appropriate habitat-specific acoustic settlement cues (Stanley et al., 2010). Since exceeding temporal thresholds are believed to be important in determining survival and subsequent juvenile development (Stanley et al. 2010), investigating the juvenile growth rates, feeding behaviours and overall mortality following the metamorphosis of turbine sound treatment megalopae would also provide insight into the possible long-term ecological effects from turbine sound. Longer-term experiments would also help to establish if these crabs are capable of habituating to the anthropogenic sound.

The absence of a difference in the median TTM between tidal turbine sound intensity treatments (i.e., 125 versus 145 dB re 1 µPa, expt. 7) suggests the observed delayed metamorphosis responses are more likely due to frequency composition of the anthropogenic sound rather than intensity alone, or at least a combination of both. The source levels of both turbine sounds are significantly greater than ambient sound and most of the acoustic energy resides in frequencies below 1 kHz (Wahlberg & Westerberg 2005) and 8 kHz (Parvin et al. 2005) in wind and tidal turbine sounds, respectively. Several peak intensities are exhibited in tidal turbine sound at 0.3, 0.8, 2 and 5 kHz (Parvin et al. 2005), while the wind turbine has a more even spread of intensity across frequencies. While it is tempting to speculate on differences in the spectra between the sounds of natural habitat, which induced a metamorphosis response, versus the turbine sounds which inhibited the response, the determination of these differences will be challenging.
Chapter Four

Characterising natural underwater sound of biological origin from a proposed tidal turbine site, the Kaipara Harbour

4.1 Introduction

The environmental impacts of wind and tidal turbines deployed in coastal waters are poorly understood, despite increasing numbers of these renewable energy generation projects. Several countries are planning to develop large scale turbine farms in coastal habitats, particularly in estuaries (Pelc & Fujita 2002; Dolman 2007). The Kaipara Harbour is the largest estuary in the southern hemisphere and has an area of ~ 947 km$^2$ and has 900 km of coastline (Heath 1975; Sim-Smith et al. 2012). The harbour is also relatively remote with little vessel activity and other underwater anthropogenic sound sources. Recently, it has become a proposed site for up to 200 tidal turbines (Argo 2006). During a spring tide, currents within the estuary can reach 2 m s$^{-1}$ with a tidal exchange of approximately $1.99 \times 10^9$ m$^3$ of water (Argo 2006). The sound levels produced by tidal turbines are estimated to be of 166 – 175 dB re 1 µPa @ 1 m (Lloyd et al. 2011) at frequencies ranging from 100 to 8000 Hz under a maximum tidal flow of 3 m s$^{-1}$ (Parvin et al. 2005; Pine et al. 2012). Commonly identified potential impacts from loud anthropogenic sound is habitat loss and exclusion, hearing loss, acoustic masking, disruption of behaviours and increased stress in marine animals (Dolman 2007; Inger et al. 2009; Slabbekoorn et al. 2010; Pine et al. 2012). It is known that natural underwater sound is an important orientation and settlement cue for a wide range of ecologically important fishes (Simpson et al. 2005; Montgomery et al. 2006; Mann et al. 2007; Radford et al. 2011; Simpson et al. 2011), coral (Vermeij et al. 2010) and crustaceans (Jeffs et al. 2003, 2005; Radford et al. 2007; Stanley et al. 2010; Pine et al. 2012).

Temperate and tropical reefs have shown patterns of periodic increases in sound levels during sunset and sunrise which are referred to as dawn and dusk choruses (Radford et al. 2008a).
During choruses, the intensity of ambient sound can increase by as much as 30 dB re 1 µPa which is due to increased crepuscular activity of many reef inhabitants (Cato 1978b). Seasonal variability of reef sound has also been found with the greatest sound intensities occurring during summer months (Radford et al. 2008a). Lunar variation in sound intensity has also been observed in reef habitats, with higher mean sound levels at 0.7 – 15 kHz recorded over new moon periods across all seasons within a temperate reef habitat (Radford et al. 2008a). Wind and waves produce frequencies in the range 10 – 10000 Hz (Wenz 1962, Radford et al. 2010) while the biological sounds from fishes and invertebrates engaged in foraging, mating, communication or territorial defence cover a wide range of frequencies (Cato 1992; Radford et al. 2008a). Ambient sounds also vary in intensity and spectral composition between reef habitats (Radford et al. 2010) and these differences are known to be important for nocturnal orientation of many organisms to specific habitats (Simpson et al. 2008a; Radford et al. 2011). Differences in underwater sound among habitats appear to be related to differences in resident soniferous animals. For example, reefs dominated by sea urchins produce sound that is more intense within biologically important frequencies (800 – 2500 Hz) compared to macro-algae dominated reefs and beach habitats (Radford et al. 2010). Snapping shrimp have also been found to likely produce sound within 2000 – 12000 Hz frequency range (Au & Banks 1998; Radford et al. 2008a) and is the most ubiquitous source of biological sound within hard and soft shore habitats in Australian and New Zealand waters (Cato 2002; Radford et al. 2008a).

However, relatively little is known about the characteristics of underwater sound in estuarine habitats. Characterising sound in estuarine habitats is important for the consideration of potential impacts of underwater sound from the placement of renewable energy generation devices in this environment. The sound emitted from tidal turbines themselves must also be considered relative to the ambient background sound of the site to help determine the potential spatial scale of impacts (Lloyd et al. 2011) and therefore, understanding the temporal and spatial variation of underwater ambient sound within estuaries is essential for assessing the potential impact of turbine sound. This study investigates the ambient sound of biological origin from within two habitats (subtidal mudflat and seagrass-dominated) in the proposed tidal turbine site, the Kaipara Harbour.
4.2 Methods

4.2.1 Recording sites

To characterise the ambient underwater sound of two habitats, underwater recordings of ambient sound were made at quarter and new moon phases during all seasons from February 2012 to February 2013. These two lunar phases were selected because there were two quarter moons per month and weather conditions during new moon periods were consistently better compared to full moon periods. The selected habitat types were subtidal mudflat (S 36° 24.608’ E 174° 22.682’; largely inhabited by the Asian mussel, *Musculista senhousia*) and seagrass–dominated (*Zostera muelleri*) (S 36° 25.964’ E 174° 18.698) habitat and selection was based on being the two primary habitat types in the Kaipara Harbour. To determine if there were differences in the sound levels and spectral composition of the underwater sound within each habitat type, two hydrophones (100 m apart) were deployed in each of the two habitats during the new moon in summer. If the mean sound levels and frequency composition between both hydrophones within the same habitat were similar, then only one recording site was used per habitat to avoid sampling redundancy thereafter.

4.2.2 Underwater sound recording system

Ambient recordings were made using up to four calibrated High Tech, Inc. HTI-96-MIN omnidirectional hydrophones (10 Hz to 60 kHz flat response), each connected to a separate watertight recording unit. The recording apparatus (containing the hydrophone, battery, recorder [20 dB gain, 16 bit, 48 kHz sampling rate] was bolted to a steel stand anchored by an iron bar and submerged in 3.4 and 3.8 m of water at low mean water spring tide in the seagrass-dominated and mudflat habitats, respectively. The marker float was anchored more than 10 m from the recording apparatus to remove any extraneous sound from the float or rope. Recording was continuous and lasted for a maximum three days.

4.2.3 Calibration

Digital recorders (EDIROL-R09HR) were calibrated for a gain setting of 20 dB before each recording by taking a series of 1 kHz pure tone sine wave recordings produced by a signal generator (Incorporate Electronics Corporation, model F34 function generator). The voltage
(\text{V}_{\text{rms}}) \text{ of the recorded } 1 \text{ kHz tone was measured and compared with the hydrophone sensitivity of } -164.6 \text{ dB re } 1 \text{ V} / 1 \mu \text{Pa. From the data, a correction factor was calculated and used within MATLAB with codes specifically written for these recordings for sound pressure measurements.}

### 4.2.4 Conditions during recording

Variability in wind speed between recording events may influence the ambient sound due to water movement at the sea surface (Radford et al. 2008a, 2010) due to the entrapment of bubbles at the surface. Therefore, average 3 h wind speeds during dawn, noon and dusk for all sound recording events were logged (average speeds recorded every 5 min during each 3 h) simultaneously at the closest weather station 15 km away at Kaipara Flats (S 36° 31.448’ E 174° 28.568’). Ambient sound recordings were made in slight sea conditions, where swell was minimal (< 1 m) and wind speeds were below 15 km h\(^{-1}\) (i.e., a maximum Beaufort scale of 3). Sea surface temperature (SST) and salinity were recorded every 15 min with StarOddi DST-CT loggers deployed with the hydrophones.

### 4.2.5 Data analysis

Mean wind speeds were compared using ANOVA for factors Seasons × Moon Phase × Diurnal Period. Salinity and SST was compared using ANOVA for factors Habitat × Seasons × Moon Phase × Diurnal Period.

Differences between recording sites within each habitat were tested using ANOVA for factors Diurnal Period × Mean Sound Level between hydrophone data from within each habitat. Spectral composition between recordings within each habitat was compared using ANOVA for factors Diurnal Period × Octave Band for sound intensity in each octave band.

Each sound recording was examined for uniformity of frequencies (to ensure no artifacts, such as extraneous sounds from wind and waves or vessel sound, were present) and ten randomly selected 10-sec samples from each diurnal period were then analysed and plotted as power spectra and octave bands (centre frequencies of 125, 200, 500, 1000, 2000, 4000, 8000, and 16000 Hz and respectively referred to as 1 – 8 octave band). Octave band plots were used as they are thought to approximate the critical hearing bands in fish (Ghazali
2011). Each subsample was high-pass filtered to 100 Hz to remove any extraneous sound from surface waves (Radford et al. 2008a; Radford et al. 2010). Data were analysed for four seasons (summer, autumn, winter and spring), two lunar phases (quarter and new moon) and four diurnal periods (dawn, noon, dusk and night). Each diurnal period was defined using published solar information (http://www.timeanddate.com). Dawn and dusk was determined by sunrise and sunset times, respectively, while noon was determined by the solar noon (the time at which the angle between magnetic north and the sun was $0^\circ$). Recordings which were to be analysed were taken from 10 min either side of sunrise, solar noon, and sunset. The night period was based on data obtained at midnight New Zealand Standard Time. All recordings were analysed using Adobe Audition and MATLAB software. Mean sound levels for 0.1 – 24 kHz were compared using ANOVA for factors Habitat $\times$ Season $\times$ Moon Phase $\times$ Diurnal Period. Octave bands within each habitat were compared using ANOVA for factors Season $\times$ Moon Phase $\times$ Diurnal Period $\times$ Octave Band. For all statistical comparisons, a $P$ value of $\leq 0.05$ was considered significant. Tukey HSD pairwise comparison tests were used to determine differences in the mean sound levels or octave bands between individual pairs of recordings where the overall interaction was found to be significant.
4.3 Results

4.3.1 Recording conditions

Wind speeds were similar across all recording conditions (Interaction ANOVA, $F_{9,409} = 0.73$, $P > 0.05$), therefore, differences between intensities and frequency composition for ambient sounds were not attributed to wind. Sea surface temperatures were significantly higher ($F_{3,1860} = 11926.58$, $P < 0.05$) during summer ($22.0 \pm 0.05^\circ C$) than autumn ($19.3 \pm 0.02^\circ C$), winter ($11.8 \pm 0.05^\circ C$) and spring ($14.5 \pm 0.03^\circ C$). Salinity was also significantly higher ($F_{3,1095} = 100.02$, $P < 0.05$) during summer ($29.6 \pm 0.01$ ppt) than autumn ($28.0 \pm 0.19$ ppt), winter ($26.3 \pm 0.19$ ppt) and spring ($27.6 \pm 0.07$ ppt).

Overall mean sound levels between the two recording sites within each habitat type were similar for both the seagrass-dominated ($F_{3,79} = 0.37$, $P > 0.05$) and mudflat ($F_{3,79} = 1.12$, $P > 0.05$) habitats (Figure 4.3.1). Sound intensities within each octave band were also similar between sites within the seagrass-dominated ($F_{7,159} = 0.11$, $P > 0.05$) and mudflat ($F_{7,159} = 1.63$, $P > 0.05$) habitats (Figure 4.3.1). Therefore, subsequent recordings (i.e., from quarter moon in summer to spring) were made from only one site within each habitat.

4.3.2 Spectral plot characteristics

The spectral composition of the seagrass-dominated habitat differed to the mudflat habitat type over a 24 h period during summer. Within the seagrass-dominated habitat only, there was a rise in spectral power in the 2 – 4 kHz bandwidth during the dawn (0600 hrs) and dusk (2030hrs) diurnal periods (Figure 4.3.2). Within the mudflat habitat, there was a rise in the 0.9 – 1.2 kHz bandwidth, and this rise correlated with a small depression in the 0.9 – 1.2 kHz bandwidth from within the seagrass-dominated habitat (Figure 4.3.2). During dawn and dusk diurnal periods within the mudflat habitat, there was a rise in the 6 – 12 kHz bandwidth (Figure 4.3.2). Spectral power in low frequencies (< 0.9 kHz) was greater in the seagrass-dominated habitat compared to the mudflat habitat.
Figure 4.3.1: Mean sound levels for 0.1 – 24 kHz (A, C) and octave band plots (B, D) for mudflat (A, B) and seagrass-dominated (C, D) habitats between two recording sites within each habitat.
Figure 4.3.2: Spectral plots of recordings from both the seagrass-dominated (blue line) and mudflat (black line) habitats at summer new moon: (A) 0000 hrs; (B) 0300 hrs; (C) 0600 hrs; (D) 0900 hrs; (E) 1200 hrs; (F) 1500 hrs; (G) 1800 hrs, and; (H) 2100 hrs.
Figure 4.3.3: Spectral plots showing the 5th, 25th, 50th, 75th and 95th percentile of recordings (1.5 hrs from at least 2 new moon phases per season) from both the mudflat (A-D) and seagrass dominated (E-H) habitats at summer (A,E), autumn (B,F), winter (C,G) and spring (D,H) new moon.
4.3.3 Octave band plots

4.3.3.1 Seagrass-dominated habitat

Sound levels within the seagrass-dominated habitat varied significantly among seasons ($F_{3,640}=467.69$, $P<0.05$), moon phases ($F_{1,640}=227.396$, $P<0.05$), diurnal periods ($F_{1,640}=42.505$, $P<0.05$) and octave bands ($F_{1,640}=2349.91$, $P<0.05$). Overall, there was a significant interaction between Season × Moon Phase × Diurnal Period × Octave Band ($F_{21,640}=56.25$, $P<0.05$) (Table 4.3.1). Sound intensity within octave bands 5, 6 and 7 during dusk increased significantly by 2 – 5 and 3 – 8 % compared to noon in autumn and winter, respectively (Figure 4.3.4). For octave bands 1, 2, 3 and 4, there was a significant decrease of 13 – 15 and 10 – 21 % in sound intensity during dusk compared to noon periods in autumn and winter, respectively. However, there was a small intensity rise in octave bands 1, 2 and 3 by 8 – 12 and 5 – 15 % during a new moon in summer and spring, respectively, where SST was highest. The largest rises in sound intensity were observed within octave band 5 and the least intensity was recorded in octave band 1 during all recording events during all seasons.

4.3.3.1 Mudflat habitat

Sound levels within the mudflat habitat varied significantly among seasons ($F_{3,640}=171.94$, $P<0.05$), moon phases ($F_{1,640}=5.97$, $P<0.05$), diurnal periods ($F_{1,640}=15.45$, $P<0.05$) and octave bands ($F_{1,640}=1143.11$, $P<0.05$). There was an overall significant interaction between Season × Moon Phase × Diurnal Period × Octave Band ($F_{21,640}=8.62$, $P<0.05$) (Table 4.3.2). Sound intensity within octave bands 6 and 7 during dusk increased significantly by 2 – 5 and 2 – 8 % compared to noon in summer and winter, respectively (Figure 4.3.5). For octave bands 1, 2, 3 and 4, there was a significant decrease by 7 – 19 % in sound intensity during dusk compared to noon periods in autumn. However, there was a small intensity rise in octave bands 1, 2, and 3 by 4 – 7 and 7 – 19 % during a new moon in summer and spring, respectively, at which time the SST was at its highest. The largest rises in sound intensity were observed within octave band 5 and the least intensity was recorded in octave band 1 during all recording events during all seasons.
Table 4.3.1: ANOVA table showing the different combinations of factors: Season, Moon Phase, Diurnal Period and Octave Band for sound from within the seagrass-dominated habitat.

<table>
<thead>
<tr>
<th>Source of variation</th>
<th>df</th>
<th>SS</th>
<th>MS</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Season</td>
<td>3</td>
<td>10.52</td>
<td>3.51</td>
<td>467.69</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Moon Phase</td>
<td>1</td>
<td>1.70</td>
<td>1.70</td>
<td>227.40</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Diurnal Period</td>
<td>1</td>
<td>0.32</td>
<td>0.32</td>
<td>42.51</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Octave Band</td>
<td>7</td>
<td>123.28</td>
<td>17.61</td>
<td>2349.91</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Moon Phase</td>
<td>3</td>
<td>4.15</td>
<td>1.38</td>
<td>186.48</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Diurnal Period</td>
<td>3</td>
<td>0.78</td>
<td>0.26</td>
<td>34.52</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Octave Band</td>
<td>21</td>
<td>7.74</td>
<td>0.37</td>
<td>49.18</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Moon Phase × Diurnal Period</td>
<td>1</td>
<td>0.52</td>
<td>0.52</td>
<td>69.16</td>
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</tr>
<tr>
<td>Moon Phase × Octave Band</td>
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<td>0.31</td>
<td>0.04</td>
<td>5.85</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Diurnal Period × Octave Band</td>
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<td>1.57</td>
<td>0.22</td>
<td>29.84</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Diurnal Period</td>
<td>3</td>
<td>8.25</td>
<td>2.75</td>
<td>367.00</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Octave Band</td>
<td>21</td>
<td>6.83</td>
<td>0.33</td>
<td>43.39</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Diurnal Period × Octave Band</td>
<td>21</td>
<td>0.84</td>
<td>0.04</td>
<td>5.35</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Moon Phase × Diurnal Period × Octave Band</td>
<td>7</td>
<td>0.37</td>
<td>0.05</td>
<td>6.82</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Diurnal Period × Octave Band</td>
<td>21</td>
<td>8.88</td>
<td>0.42</td>
<td>56.25</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Total</td>
<td>640</td>
<td>62258.86</td>
<td></td>
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<td></td>
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</table>

Table 4.3.2: ANOVA table showing the different combinations of factors: Season, Moon Phase, Diurnal Period and Octave Band, for sound from within the mudflat habitat.

<table>
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<tr>
<th>Source of variation</th>
<th>df</th>
<th>SS</th>
<th>MS</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Season</td>
<td>3</td>
<td>7.03</td>
<td>2.35</td>
<td>171.94</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase</td>
<td>1</td>
<td>0.08</td>
<td>0.08</td>
<td>5.96</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Diurnal Period</td>
<td>1</td>
<td>0.21</td>
<td>0.21</td>
<td>15.45</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Octave Band</td>
<td>7</td>
<td>109.12</td>
<td>15.59</td>
<td>1143.11</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Moon Phase</td>
<td>3</td>
<td>6.97</td>
<td>2.32</td>
<td>170.45</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Diurnal Period</td>
<td>3</td>
<td>5.01</td>
<td>1.67</td>
<td>122.54</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Octave Band</td>
<td>21</td>
<td>7.30</td>
<td>0.35</td>
<td>25.50</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase × Diurnal Period</td>
<td>1</td>
<td>1.31</td>
<td>1.31</td>
<td>96.21</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase × Octave Band</td>
<td>7</td>
<td>0.67</td>
<td>0.10</td>
<td>7.04</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Diurnal Period × Octave Band</td>
<td>7</td>
<td>4.31</td>
<td>0.62</td>
<td>45.18</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Diurnal Period</td>
<td>3</td>
<td>4.54</td>
<td>1.51</td>
<td>111.05</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Octave Band</td>
<td>21</td>
<td>5.27</td>
<td>0.25</td>
<td>18.39</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Diurnal Period × Octave Band</td>
<td>21</td>
<td>1.88</td>
<td>0.09</td>
<td>6.55</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase × Diurnal Period × Octave Band</td>
<td>7</td>
<td>0.75</td>
<td>0.10</td>
<td>7.81</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Moon Phase × Diurnal Period × Octave Band</td>
<td>21</td>
<td>2.47</td>
<td>0.12</td>
<td>8.62</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Total</td>
<td>640</td>
<td>60202.12</td>
<td></td>
<td></td>
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</table>
Figure 4.3.4: Octave band level plots comparing noon and dusk recordings from the seagrass-dominated habitat over all seasons during a quarter moon (QM) and new moon (NM) phases.
Figure 4.3.5: Octave band level plots comparing noon and dusk recordings from the mudflat habitat over all seasons during a quarter moon (QM) and new moon (NM) phase.
4.3.4 Mean sound levels

4.3.4.1. Between habitat types

Mean ambient sound levels varied significantly among habitats ($F_{1,640} = 250.63, P < 0.05$), seasons ($F_{3,640} = 460.63, P < 0.05$), moon phases ($F_{1,640} = 8.58, P < 0.05$), and diurnal periods ($F_{3,640} = 73.03, P < 0.05$) (Table 4.3.3; Figure 4.3.6). However, mean sound levels were similar between Habitat × Moon Phase × Diurnal Period ($F_{3,640} = 2.33, P > 0.05$) (Table 4.3.3). There was a significant overall interaction in mean sound levels between Habitat × Season × Moon Phase × Diurnal Period ($F_{9,640} = 12.30, P < 0.05$) (Table 4.3.3; Figure 4.3.6). Mean sound levels of underwater sound were similar between summer and spring ($P > 0.05$) (Figure 4.3.6). Mean sound levels were greatest at dawn periods for summer (112 – 113 dB re 1 μPa) and autumn (116 – 123 dB re 1 μPa) in both habitats, followed by dusk (114 dB re 1 μPa), midnight (112 dB re 1 μPa) and noon (109 dB re 1 μPa). However, for winter and spring, mean sound levels were greater during both dawn and dusk (112 dB re 1 μPa) periods, followed by midnight (108 dB re 1 μPa) and noon (108 dB re 1 μPa). Mean sound levels from the seagrass-dominated habitat were generally higher compared to the mudflat habitat.

4.3.4.2 Seagrass-dominated habitat

The highest mean sound level (123 dB re 1 μPa) was recorded at dawn during the new moon in autumn and was significantly higher by 10 % than sound levels during the quarter and new moon phases over winter, spring and summer. The lowest mean sound level (106 dB re 1 μPa) was recorded at noon during the new moon in spring. For summer, autumn and winter, mean sound levels were greater during the new moon period than their respective quarter moon periods. Differences in mean sound levels between quarter and new moon periods were greatest in autumn (10.4 dB re 1 μPa) and the smallest difference between the two moon phases was recorded in spring (0.7 dB re 1 μPa).
4.3.4.3 Mudflat habitat

The highest mean sound level (116 dB re 1 µPa) was recorded at dawn during the new moon in autumn and was significantly higher by 7 % than sound levels during the quarter and new moon phases over winter, spring and summer. The lowest mean sound level (101 dB re 1 µPa) was recorded at noon during the quarter moon in winter. For autumn, winter and spring, mean sound levels were greater during the new moon period than their respective quarter moon periods. Differences in mean sound levels between quarter and new moon periods were greatest in autumn (8.4 dB re 1 µPa) and the smallest difference between the two moon phases was recorded in summer (2.0 dB re 1 µPa).

Table 4.3.3: ANOVA table showing the different combinations of factors: Habitat, Season, Moon Phase, and Diurnal Period, for Mean Sound Levels for 0.1 – 24 kHz.

<table>
<thead>
<tr>
<th>Source of variation</th>
<th>df</th>
<th>SS</th>
<th>MS</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat</td>
<td>1</td>
<td>1289.82</td>
<td>1289.82</td>
<td>250.63</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season</td>
<td>3</td>
<td>7111.64</td>
<td>2370.55</td>
<td>460.63</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase</td>
<td>1</td>
<td>44.15</td>
<td>44.15</td>
<td>8.58</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Diurnal Period</td>
<td>3</td>
<td>1127.45</td>
<td>375.82</td>
<td>73.03</td>
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</tr>
<tr>
<td>Habitat × Season</td>
<td>3</td>
<td>1013.31</td>
<td>337.77</td>
<td>65.63</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Habitat × Moon Phase</td>
<td>1</td>
<td>17.51</td>
<td>17.51</td>
<td>3.40</td>
<td>&gt; 0.05</td>
</tr>
<tr>
<td>Habitat × Diurnal Period</td>
<td>3</td>
<td>66.35</td>
<td>22.12</td>
<td>4.30</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Moon Phase</td>
<td>3</td>
<td>556.48</td>
<td>185.49</td>
<td>36.04</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Season × Diurnal Period</td>
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<td>336.47</td>
<td>37.39</td>
<td>7.20</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Moon Phase × Diurnal Period</td>
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<td>132.45</td>
<td>44.15</td>
<td>8.58</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Habitat × Season × Moon Phase</td>
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<td>476.73</td>
<td>158.91</td>
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</tr>
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<td>Habitat × Season × Diurnal Period</td>
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<td>487.52</td>
<td>54.17</td>
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<td>Habitat × Moon Phase × Diurnal Period</td>
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<td>36.02</td>
<td>12.01</td>
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<td>&gt; 0.05</td>
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<td>Season × Moon Phase × Diurnal Period</td>
<td>9</td>
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</tr>
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<td>110.72</td>
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<td>2.39</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>Total</td>
<td>640</td>
<td>7986595.69</td>
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</table>
Figure 4.3.6: Mean sound levels for 0.1 – 24 kHz at each diurnal period for seagrass-dominated and mudflat habitats during the quarter moon (QM) and new moon (NM) for summer, autumn, winter and spring.
4.4 Discussion

Understanding the temporal and spatial variability of ambient underwater sound from a proposed tidal turbine site is a fundamental component of the environmental impact assessment. Ambient sound from temperate and tropical reefs has been well studied (Cato 1992, 2002; Radford et al. 2008a; Radford et al. 2010), with ambient sound significantly differing between habitats types, such as a beach, macro-algal dominated reef and an urchin barren (Radford et al. 2010). This study aimed to characterise the ambient sound of biological origin from within two major habitats present in a large temperate estuary, the Kaipara Harbour.

Ambient underwater sound showed significant seasonal, lunar and diurnal variability both within and between the two different habitats within the harbour. Within both habitats, there was an increase in mean sound levels between night and dawn during the new moon across all seasons. Dawn and dusk choruses are defined as marked increases in the combined ambient sound levels due to higher crepuscular activity from inhabiting animals during dawn and dusk (Radford et al. 2008a). Within both habitats, dawn or dusk choruses were observed in all seasons. However, within the seagrass-dominated habitat, choruses were not consistent across the lunar month in summer or spring, which differed from the mudflat habitat where dawn or dusk choruses were not consistent across the lunar month in autumn and spring. This finding differed to that found from reef habitats, which showed consistent presence of both dawn and dusk choruses all year-round (Radford et al. 2008a). The absence of either dawn or dusk choruses over 24 h is not well understood, as it has not been reported for any other coastal habitat. The seasonal and lunar variation in sound levels could likely be linked to plankton blooms, nutrient fluxes, and/or annual recruitment patterns of juvenile fishes, such as the snapper (*Chrysophrys auratus*), during spring and summer (Sim-Smith et al. 2012). Similar to reef habitats, the greatest differences in sound production between the two sampled habitats in this harbour occurred during dusk, which suggests that dusk is the better time to investigate differences in the ambient sound of different habitats. This same pattern has also been noted for urchin barrens, macro-algal dominated reefs, and beach habitats and appears likely to be related to the crepuscular activity patterns of resident soniferous organisms (Radford et al. 2010).
The current study has shown that within the seagrass-dominated habitat, the highest sound intensity was generally within octave bands 5 and 6, which equates to a rise in sound intensity between 2 – 4 kHz. This contrasted to the mudflat habitat, where the greatest sound intensity was generally found within octave band 7, i.e., 6 to 12 kHz. From noon to dusk, the sound intensity within octave bands 5, 6 or 7 noticeably increased in autumn, winter and spring within the seagrass-dominated habitat and in summer and winter within the mudflat habitat. The potential sources of the predominant frequency ranges found from the seagrass-dominated habitat are unknown, because fish are known to generally produce sounds at frequencies lower than 1 kHz (Ghazali 2011; Parmentier et al. 2011; Tellechea et al. 2011). One of the most ubiquitous sound sources of biological sound within either habitat were snapping shrimp (Alpheus sp. and Synalpheus sp.), which collectively produce a distinct crackling sound within 2 – 12 kHz (Cato 1976, 1978a, 1992; Radford et al. 2008a) and are likely to be responsible for the higher intensities in octave bands 5, 6 and 7.

By identifying which animals are producing the predominant frequencies within each habitat could lead to a useful method for remotely assessing biodiversity for environmental monitoring and conservation management, in much the same way as indicator species are used in freshwater ecosystems (Walsh et al. 1982; Sawyer & Bodammer 1983). The use of underwater acoustics as a remote tool to measure a particular aspect of an ecosystem has been recently demonstrated for the monitoring of the canopy height and spatial coverage distribution in seagrass meadows (Paul et al. 2011). Therefore, understanding other biological aspects of ambient sound in both reefs and estuaries could provide a remote measure for ecosystem health.

Although the majority of sound intensity within both habitats was in the higher octave bands, a rise in octave bands 1, 2 and 3 (central frequencies of 125, 200 and 500 Hz) during dusk was unique to the new moon period in spring and summer. These lower frequencies (< 1 kHz) are within the hearing thresholds of many fish and crab species (Budelmann 1992; Myrberg 1997; Kenyon et al. 1998; Popper et al. 2001) and within the frequency ranges of calls from some reef dwelling fishes (Ghazali 2011). For example, studies concerning the sound production in the threeband pennantfish, Heniochus chrysostomus, have shown calls to be harmonic and between 130 – 180 Hz (Parmentier et al. 2011) and the bicolour damselfish, Pomacentrus partitus, has been found to produce sounds around 700 – 800 Hz (Myrberg et al. 1986). The rise in lower frequencies over the new moon period in spring and summer
could correlate to increasing spawning in fishes, with higher recorded intensities below 1 kHz coinciding with settlement during summer (Jones 1980; Radford et al. 2008a; Sim-Smith et al. 2012).

This study has shown spatial variability in mean sound levels and spectral composition between two estuarine habitats in close proximity (km’s) within the Kaipara Harbour which is consistent with the results of studies of ambient sound in temperate reef habitats (Radford et al. 2010). The localised variability in ambient sound associated with mudflat and seagrass-dominated habitats may be important for mediating the settlement behaviour of larval crabs aiming to reach their preferred settlement habitats. For example, compared to silent control treatments, the post-larvae of Austrohelice crassa settled faster when exposed to estuarine mudflat sound (Pine et al. 2012), but no difference between the control treatment when exposed to reef sound (Stanley et al. 2011). The significant differences in ambient sound occurring within relatively short distances found in the current study, coupled with previous research, suggests further evidence for potential habitat selection based on ambient sound for settlement stage organisms (Radford et al. 2010; Pine et al. 2012).

Both estuarine habitats examined in this study produced sound of sufficient intensities and frequencies to be transmitted beyond the spatial extent of the habitat, and therefore the sound could potentially be an important orientation cue for settlement stage post-larvae of estuarine organisms. This has also been reported for the ambient sound from reef habitats (Radford et al. 2010). One of the potential impacts of turbine sound is the masking of ambient underwater sound whereby the anthropogenic noise “drowns out” the natural ambient sound being used as a cue by organisms. The sound from an operating ‘Seaflow’ tidal turbine has been measured to have a source level of 175 dB re 1µPa @ 1 m with peak intensities at 0.1, 0.8, 2, 5 and 8 kHz (Parvin et al. 2005) at a maximum tidal flow of 3 m s⁻¹ (Richards et al. 2007; Lloyd et al. 2011). With a lot of the sound energy of the turbine residing in low frequencies and at greater intensities than ambient estuarine sound, masking of the ambient sound levels measured in this study are likely over a relatively large area surrounding each turbine. Given the likely importance of estuarine ambient sound to estuarine larvae, acoustic surveys from within proposed tidal turbine sites is therefore recommended to properly identify and manage potential impacts of the anthropogenic noise emitted by turbines.
Chapter Five

The cumulative effect on sound levels from multiple underwater anthropogenic sound sources in shallow coastal waters

Published as:

5.1 Introduction

There is mounting evidence that natural underwater sound is important in determining the behaviour and ecology of marine organisms (Jeffs, Tolimieri & Montgomery 2003; Simpson *et al.* 2005; Montgomery *et al.* 2006; Slabbekoorn *et al.* 2010). Background underwater sound levels from anthropogenic sources within shallow coastal environments are significantly increasing in some parts of the world (Ross 1993; Mazzuca 2001; Andrew *et al.* 2002; Popper & Hastings 2009b; Popper & Hastings 2009a) and high levels of anthropogenic sound have been linked to hearing loss in fishes (McCauley, Fewtrell & Popper 2003; Smith, Kane & Popper 2004a; Smith, Kane & Popper 2004b; Popper & Hastings 2009a), delayed larval settlement in estuarine crustaceans (Pine, Jeffs & Radford 2012), habitat exclusion in marine mammals (Thomsen 2009), disruption of schooling behaviour (Sarà *et al.* 2007), increased stress responses (Smith, Kane & Popper 2004b; Wysocki, Dittami & Ladich 2006), and masking of normal acoustic communication among fishes (Vasconcelos, Amorim & Ladich 2007; Codarin *et al.* 2009). The construction and operation of sea-based wind turbines and underwater tidal turbines has increased and is rapidly becoming a source of anthropogenic underwater sound likely to impact marine life, especially acoustically sensitive species (Dolman 2007; Inger *et al.* 2009; Slabbekoorn *et al.* 2010; Halvorsen 2012). Recordings of underwater sound from operating sea-based wind turbines have been made in
Denmark, Sweden and Germany (Madsen et al. 2006). Maximum emitted sound energy is generally dominated by frequencies below 1 kHz (Koschinski et al. 2003; Madsen et al. 2006; Tougaard, Henriksen & Miller 2009), and sound pressure levels (SPL) between 109–126 dB re 1 µPa at 14 to 20 m from the sound source (Tougaard, Henriksen & Miller 2009). This sound range is within the main hearing range of many fishes and invertebrates (Budelmann 1992; Myrberg Jr 1997; Kenyon, Ladich & Yan 1998; Popper, Salmon & Horch 2001; Lovell et al. 2005). Audibility of wind turbine sound (< 500 Hz) is expected to be low for harbour porpoises beyond 70 m from the sound source, however the audibility of wind turbine sound for harbour seals is expected to be within several kilometres (Tougaard, Henriksen & Miller 2009). The detection distance of wind turbine sound in three species of fish, with different hearing capabilities, has been modelled to range between 0.4 and 25 km at wind speeds of 8 to 13 m s\(^{-1}\) (Wahlberg & Westerberg 2005). In both of these studies a single windmill emitting noise in isolation was considered as the source of the underwater sound, not an array of many wind turbine sources which is the typical format for offshore wind turbine farms. In the last five years the installation of offshore wind turbines globally has increased with the installation of nearly 4,000 MW of new production capacity, bringing overall production from offshore wind farms to over 5,000 MW (LORC 2013).

In contrast to wind turbines, there are few reported field measurements of the underwater sound emanating from operating tidal turbines, and actual recordings are rare. Field measurements of the underwater sound from a full-scale SeaFlow 0.3 MW tidal turbine found that its acoustic emissions were continuous and composed of a wide range of frequencies ranging from 100 to 8000 Hz (Parvin et al. 2005). The SeaFlow turbine had a SPL of approximately 166–175 dB re 1 µPa at 1 m under a maximum of 3 m s\(^{-1}\) tidal flow; much higher than that observed in wind turbines and hence of greater ecological concern (Lloyd, Turnock & Humphrey 2011) (albeit audibility by fishes is dependent on the signal to noise ratio (SNR) as turbine units are typically installed in high energy environments). However, source levels are likely to vary depending on turbine design and local environmental conditions at the time of measurement (Lloyd, Turnock & Humphrey 2011).

While the propagation of sound underwater is well studied, the underwater sound field from one or several operating tidal turbine units in close proximity (<100 m) is poorly understood, thereby limiting the assessment of ecological impacts. The propagation of sound from dipole sources has also been well studied (Russell, Titlow & Bemmen 1999), however, turbine
farms do not represent dipole situations as turbines are typically installed more than a few metres apart. The lack of such knowledge affects the industry, and causes concerns for the public, stakeholders and investors about the scale of ecological impacts from underwater sound generated from this activity, because sound has the ability to travel large distances underwater due to low attenuation (Halvorsen 2012). The spatial scale of the ecological impacts from the underwater sound from tidal turbines is sometimes estimated using geometric spreading models that are derived from theoretical models of wave propagation (Urick 1983; Richardson et al. 1995). However, understanding how turbine sound propagates in shallow coastal water and developing accurate propagation models is critical for adequate environmental impact assessment during the development phase of tidal energy projects. Furthermore, there is a degree of uncertainty around the current methods for the theoretical modelling of the sound field where multiple tidal turbines are within the acoustic range of one another, since wave propagation models currently accepted by some regulatory authorities are based on one sound source. Therefore, there is a need for the collection of field data to validate geometric models for the underwater sound from tidal turbines, especially for multiple turbines operating within the same acoustic field. This study investigates the propagation of underwater tidal turbine sound from both a single monopole and a twin monopole sound source and presents field measurements for comparison with estimates derived from the theoretical geometric spreading models sometimes used by regulatory bodies.
5.2 Methods

5.2.1 Tidal turbine sound file and playback

It was not possible to obtain field recordings of operational tidal turbines because operators refused to supply them for this study. Thus, a digital analogue which matched the published frequency composition (Parvin et al. 2005) and spiked intensities of a tidal turbine operating under a maximum tidal flow of 3 m s\(^{-1}\) was used for sound propagation experiments.

Sound files for playback through underwater loudspeakers were compiled using Adobe Audition acoustic software and consisted of a 10-second period of turbine noise followed by 10 seconds of silence, repeated 10 times. This file was played in a continuous loop during the experiments. For field experiments where only one loudspeaker was used (i.e. the single monopole source), the loudspeaker played back the sound file allowing the sound together with ambient background sound to be measured during the 10-second periods of broadcast tidal turbine sound and the ambient background noise was measured during the subsequent 10 seconds of silence. This facilitated subsequent comparisons of the pair of measurements and allowed the contribution of sound from the loudspeaker over and above the ambient background noise to be determined. For field experiments where two loudspeakers were used, each loudspeaker played the same tidal turbine sound file but with the 10-second periods of sound out of phase by 5 seconds. This provided a sound field with 5-second intervals where both sources were operating simultaneously, each independently, as well as a 5-second period when no sources were operating, allowing for measurement of ambient background noise.

5.2.2 Field-based sound propagation experiments

All experiments were completed between July and August 2012 in Omaha Bay, north-eastern New Zealand (36° 19.5961´ S, 174° 48.6833´ E).

To assess the propagation of turbine sound, two separate experiments were conducted. The first experiment used a single monopole and the second used a twin monopole sound source. For the single monopole experiment, an underwater loudspeaker (Lubell Lab Inc. LL964, specifications: 0.2-20 kHz, maximum output 170 dB re 1 µPa) connected to an amplifier and Phillips MP3 player contained in a watertight housing was deployed at a depth of 25 m
The loudspeaker played back tidal turbine sound at an output level of 154 dB re 1 µPa at 1 m. A 6-km long sampling transect was used that ran in a seaward direction perpendicular from the shoreline, passing the underwater loudspeaker at the 0-m recording site which was located 1000 m from the origin of the transect. A calibrated High Tech Inc. HTI-96 omnidirectional hydrophone (10 Hz to 60 kHz flat response) connected to a watertight recording housing unit (20 dB gain, 16 bit, 48 kHz sampling rate) was used to record sound at a selection of sites along the transect at 0, 100, 200, 500 and 1000 m in both directions (i.e. shoreward and seaward from the speaker), while recordings taken at 2000 and 5000 m were in the seaward direction only. At each recording site, a 7-min recording was taken with a free floating hydrophone to reduce any extraneous sound associated with recording from a vessel.

The propagation of turbine sound during the twin monopole experiment was assessed in a similar manner to the single monopole experiment and used the same sampling transect layout, however, two LL964 underwater loudspeakers were used, each deployed 65 m perpendicular to either side of the 0-m position on the transect line in a water depth of 25 m (Figure 5.2.1). The calibrated omnidirectional HTI-96 hydrophone was used to make 7-min recordings of played back turbine sound along the transect at sites 0, 100, 200, 500 and 1000 m in both directions, as well as at sites 2000 and 5000 m in the seaward direction. Another calibrated omnidirectional HTI-96 hydrophone in a watertight recording unit was placed at the 0-m site and made continuous recordings while remaining stationary 1 m above the seafloor throughout the twin monopole experiment to confirm adequate playback from both sources at all times.

Recording was ceased when any operating vessels were visible. During both the single monopole and twin monopole experiments StarOddi CTD loggers simultaneously recorded sea surface temperature and salinity every 30 sec over 30 min to confirm there was no difference between the two experiments. The absence of a thermocline, which can lead to acoustic channelling, was also confirmed by measuring the temperature and salinity 1 m above the sea floor beneath the speaker(s) and comparing to the sea surface measurements. The experiments took place during calm sea conditions (< 1 m swell) and less than 5 m s⁻¹ winds.
Figure 5.2.1: Study site in Omaha Bay along the northeastern coast of New Zealand. Squares represent speaker(s) location, while the arrow shows the direction and location of the transect line along which recordings were taken on.
\section*{5.2.3 Data analyses}

Each of the 7-min sound recordings from both single and twin monopole experiments were randomly divided into five 1-min samples; each sample containing three 10-second (single monopole experiment) or 5-second (twin monopole experiment) periods of turbine sound and three 10-sec (single monopole experiment) or 5-sec (twin monopole experiment) periods of ambient background noise. Recordings were then bandpass filtered for 100–24000, 100–800, 801–2000 and 2001–20000 Hz frequency bands to calculate measured sound levels for each. These frequency bands were selected because the underwater sound produced by most turbines contains the majority of sound energy in the frequencies below 2 kHz, which is also the hearing frequency range of many fish (Halvorsen 2012), and is the same frequency range as produced by feeding sea urchins (Radford et al. 2010). Ambient sound levels at each recording site for both experiments were also measured and used to calculate the contribution of turbine sound over ambient dB levels for each recording site using MATLAB and Adobe Audition acoustic software. Multiple linear regression analysis (mlrANOVA) was used to compare graphical slopes of turbine and ambient sound levels from recordings in each of the four frequency bands at each distance from the loudspeaker. A Split-Plot ANOVA was used to directly compare mean sound levels at each of the six distances from the loudspeaker, with the assumption that variation between days was negligible. Homogeneity of slope for multiple regression lines was tested by evaluating the F statistic for pooled regressions (Zar 2010). Differences in surface versus seafloor temperatures or salinity, as well as surface temperatures or salinity, between the single monopole and twin monopole experiments were tested using t-tests after confirmation of assumptions of normality and homogeneity of variances for the source data. For all tests, a $P$ value of $\leq 0.05$ was considered significant. Where an overall experimental effect had been detected, Holm-Sidak pairwise comparison tests were used to isolate the mean sound levels which differed from the others. All statistical analyses were carried out using the statistical software Sigma Plot 11.0 and SPSS 17.0.

The theoretical propagation of turbine sound as modelled by cylindrical (calculated by $10 \log r$, where $r$ is the distance (m) from the source) and spherical (calculated by $20 \log r$) geometric spreading was also used to estimate sound levels for 0.1–20 kHz for each of the equivalent transect sites so they could be compared to field measurements from each experiment. When more monopoles are added at equal distances
(greater than the wavelength) the overall source level can be calculated by adding the intensity of each monopole in Watts per m² and converting to decibels (Urone 2001; Nelson 2007). This assumes the receiver is equidistant from the source and the depths are indifferent along both propagation paths between the receiver and sound sources. The overall intensity at the receiver doubles in the twin monopole experiment and in this instance correlates to an increase of 3 dB re 1 µPa. Therefore, in the twin monopole experiment the sound source was treated as a single monopole source plus an estimated 3 dB re 1 µPa to account for the contribution of the second monopole source. A significant difference between the measured mean sound level and the theoretical estimate was determined using a normal distribution Z test ($P < 0.05$).
5.3 Results

5.3.1 Confirmation of sound sources

The output of tidal turbine sound by the underwater loudspeakers showed a similar spectral composition and matched key frequency peaks to the original source signal (Figure 5.3.1) (Parvin et al. 2005).

5.3.2 Conditions during recording

There was no significant difference in sea surface temperature (approximately 14.8 ± 0.04 °C) ($T_{0.05, 54} = 0.76, P = 0.45$) or salinity (36.1 ± 0.05 ppt) ($T_{0.05, 54} = 1.40, P = 0.17$) between all recording events. The absence of a thermocline was also confirmed in the single monopole ($T_{0.05, 26} = 1.18, P = 0.25$) and twin monopole ($T_{0.05, 26} = 1.07, P = 0.30$) experiments.

5.3.3 Propagation of tidal turbine sound

No significant differences were found among the mean ambient sound levels (0.1 – 20 kHz) from all recording sites within the single monopole (mlrANOVA, $F_{5,29} = 0.62, P > 0.05$) and twin monopole (mlrANOVA, $F_{5,29} = 1.59, P > 0.05$) experiments, as well as between experiments (mlrANOVA, $F_{5,59} = 0.98, P > 0.05$). Underwater sound levels for the combined turbine and ambient sound decreased with the distance from the source(s) in both the single monopole ($R = 0.82, F_{1,29} = 58.2, P < 0.001$) and twin monopole ($R = 0.88, F_{1,29} = 98.1, P < 0.001$) experiments (Figure 5.3.2, Figure 5.3.3). At 100 m seaward from the loudspeaker in the single monopole experiment, the measured contribution of turbine sound (over the ambient dB level) to the overall sound level (136 ± 0.1 dB re 1 µPa @ 100 m) was 24.5 ± 0.1 dB re 1 µPa and this decreased to 12.3 ± 0.28 dB re 1 µPa at 1 km, and 6.7 ± 0.17 dB re 1 µPa at 5 km (Figure 5.3.2). The measured sound levels in the twin monopole experiment differed significantly in that at 100 m the measured contribution of turbine sound to the overall sound level (141 ± 0.1 dB re 1 µPa @ 100 m) was 28.8 dB re 1 µPa ($F_{4,144} = 14.3, P < 0.01$). The decrease in sound energy at 0.1–20 kHz with increasing distance from the source(s) out to 5 km was smaller in the single monopole experiment compared with the twin monopole experiment ($F_{5,48} = 118.5, P < 0.01$).
Figure 5.3.1: Spectrograms of the experimental turbine sound source as a digital source file generated from (Parvin et al. 2005) (black trace) and when played back through an underwater loudspeaker (Lubell LL964) in seawater and recorded at 1 m at an equivalent overall sound level (153 dB re 1 µPa @ 1m) (grey trace).
5.3.4 Sound propagation in different frequency bands

At all distances from the sound source(s), the intensity of the sound for all frequency bands was consistently greater in the twin monopole experiment compared to the single monopole experiment due to the cumulative effect of the twin sound sources. There was a smaller decrease in sound energy with distance from the source(s) for the of 100–800 Hz frequency band compared to frequencies above 2000 Hz in both the single monopole (mlrANOVA, F_{4,144} = 44.7, P < 0.001) and twin monopole (mlrANOVA, F_{4,129} = 48.309, P < 0.001) experiments (Figure 5.3.2, Figure 5.3.4). The measured decrease in intensity with distance from the source of the low frequency sound (100–800 Hz) was less in the single monopole experiment (decrease of 7.5 ± 0.25 dB at 5 km) compared to the same frequency band in the twin monopole experiment (decrease of 8.2 ± 0.14 dB at 5 km) (mlrANOVA, F_{2,56} = 37.9, P < 0.01). There was a decrease in the sound level over 5 km of 13.6 ± 0.28 dB in the 801–2000 Hz frequency band from the single monopole experiment (mlrANOVA, F_{2,56} = 21.4, P < 0.05), compared to 17.0 ± 0.32 dB in the twin monopole experiment. The decrease in sound level with increasing distance from the source in the highest frequency band (i.e. 2001–20000 Hz) was greater in the twin monopole experiment than for the single monopole experiment (mlrANOVA, F_{2,56} = 51.4, P < 0.01) (Figure 5.3.2).

5.3.5 Sound propagation as per cylindrical and spherical models

Estimated sound levels from cylindrical and spherical geometric spreading of a hypothetical single and twin monopole sound source of the same source intensity as the speaker differed from the single monopole and twin monopole field measurements for all recording sites (P < 0.05) (Figure 5.3.5). For example, at 1 km the estimated sound levels for cylindrical spreading of sound of 0.1–20 kHz were 3.7 and 6.4 dB less than the mean measured level in the single monopole and twin monopole experiments, respectively. Furthermore, estimates based on the spherical spreading of sound had even greater divergence from field measurements in the single monopole and twin monopole experiment, with a theoretical estimate of sound intensity at 1 km which was 33.6 and 36.4 dB below the mean field measurement in the single monopole and twin monopole experiments, respectively.
Figure 5.3.2: Recorded mean sound levels at recording sites at six set distances from the sound source(s): (A) Combined ambient and turbine sound at 0.1 – 20 kHz (Single monopole: \( y = -0.0027x + 131.06 \); Twin monopole: \( y = -0.0036x + 136.77 \)); (B) Level of turbine sound after ambient dB levels are removed within each experiment for three consecutive frequency ranges: Single monopole experiment (100 – 800 Hz: \( y = -0.0018x + 120.34 \); 801 – 2000 Hz: \( y = -0.0022x + 124.93 \); 2001 – 20000 Hz: \( y = -0.0031x + 126.39 \)); (C) Level of turbine sound after ambient dB levels are removed within each experiment for three consecutive frequency ranges: Twin monopole experiment (100 – 800 Hz: \( y = -0.0013x + 122.97 \); 801 – 2000 Hz: \( y = -0.0034x + 132.7 \); 2001 – 20000 Hz: \( y = -0.0039x + 131.86 \)).
Figure 5.3.3: Propagation of tidal turbine sound (dB re 1 μPa) from: (A) 1 sound source; (B) 2 sound sources (125 m apart); (C) 3 sound sources (125 m apart).
Figure 5.3.4: Spectrograms of the recorded tidal turbine sound with distance from the Lubell LL964 underwater loudspeaker: (A) Twin-monopole experiment; and (B) Single monopole experiment.
Figure 5.3.5: Level of turbine sound (including ambient background levels) recorded at various distances from the source for 0.1 – 20 kHz: (A) from the single monopole experiment and as estimated from conservatively modified equations (Eq. 1 and 3), as well as cylindrical and spherical geometric spreading models, and; (B) from the twin monopole experiment and as estimated from a conservatively modified equation (Eq. 2), as well as cylindrical and spherical geometric spreading models.
5.4 Discussion

As global commercial interest in marine renewable energy increases, the need for an improved understanding of the propagation of anthropogenic sound is critical for assessing the potential impacts on marine life. Recent research has shown that a number of important coastal species show sensitivity to underwater sound, including the marine larvae of crabs (Radford, Jeffs & Montgomery 2007; Stanley, Radford & Jeffs 2010; Stanley, Radford & Jeffs 2011), mussels, coral, fishes and mammals (Richardson et al. 1995; Myrberg Jr 1997; Montgomery et al. 2006; Vermeij et al. 2010; Wilkens, Stanley & Jeffs 2012), as well as cephalopods (Mooney et al. 2010). While research investigating the impacts of turbine sound on these animals is limited, two studies have found evidence of turbine sound causing low levels of tissue damage in Chinook salmon (Halvorsen 2012), and delay to the metamorphosis and subsequent settlement of estuarine crab larvae (Pine, Jeffs & Radford 2012).

The transmission of sound in shallow water is thought to be well understood, with several theoretical propagation models in general use, especially in considering potential environmental impacts from proposed placement of turbine developments in shallow coastal waters (Urick 1983; Richardson et al. 1995; Tindle & Deane 2005). However, geometric models consider conditions for a single monopole source or a single frequency band. When more sources are included, the sound field becomes more complex with the potential for interference and the cancelling of some frequencies (Lurton 2002). Compared to the single monopole experiment, the overall sound intensity produced in the twin monopole experiment was greater at all frequencies, indicating an accumulation of sound intensity from the two experimental sound sources. In New Zealand, estimated impact zones from anthropogenic sound are frequently based on the theoretical output and propagation of a single turbine as a sound source. For example, an environmental impact assessment for the installation of more than 50 tidal turbines in one harbour in New Zealand estimated the acoustic impact zone based on theoretical modelling of only one operating turbine unit (Argo 2006). However, the results of this current study indicate that actual impact zones from turbine farms may therefore be underestimated by not taking into account the accumulation of sound energy from the multiple sound sources.
The current study showed the propagation of turbine sound to vary between the single monopole and twin monopole experiments, with the sound levels from the twin monopole source decreasing more rapidly than the single monopole source with increasing distance from the source(s). This difference may be due to the cancellation caused by sound from the two sources being in counter phase (Jensen 2011).

In the current study the floating speaker(s) would have been constantly moving in the water column which could have created a sound field that is different to one radiating from a tidal turbine farm, where the sound source(s) is attached to the seafloor and stationary. There was an attempt to control for these factors by conducting experiments in calm weather, low tidal currents and minimal swell (< 1 m). The benthic attachment of a tidal turbine may also transfer additional acoustic energy which could then propagate via the seafloor and would be available for transfer into the water column at some distance from the turbine.

The underwater sound propagation models that are sometimes used to estimate sound impact zones by regulatory bodies are simple geometric models. These models rarely incorporate additional environmental variables, such as water depth and attenuation coefficients in seawater. It is generally accepted that spherical spreading of underwater sound occurs in deeper waters, where the distance between the receiver and source is less than the depth, while cylindrical spreading occurs in shallower water where the distance between the receiver and source is greater than the depth (Wahlberg & Westerberg 2005). However, in the current study, the measured decrease in sound intensity with distance was smaller than predicted by geometric modelling alone (Figure 5.3.5). The underestimation of turbine sound levels with distance by these models is most likely the result of the lack of control for other variables that can affect sound propagation, such as water depth, grazing angles, reflection, seabed propagation and frequency dependant absorption. This study has established a baseline for the propagation of turbine sound in shallow coastal waters at an output level of 154 dB re 1 µPa at 1 m over 5 km from the sound source under near-ideal conditions over a homogenous soft sediment seafloor. Regulatory bodies depend on spreading models to determine the distances over which underwater sound will propagate as a basis for assessing the potential scale of the environmental impact of underwater sound emitted from tidal turbines and other anthropogenic sources of underwater sound. Using the data from this study, a modified equation from Richardson et al. (1995) which incorporates both spherical and cylindrical spreading, a published attenuation coefficient for shallow water (Richardson et al. 1995), a
spreading constant (from field measurements in the current study) for shallow water and transmission losses due to depth and frequency dependant volumetric absorption (François & Garrison 1982a,b) could be used as a more conservative estimate for a single monopole (Eq. 1) or twin monopole (Eq. 2) sound source. This equation is given either by:

\[ SL - 15 \log RO - 10 \log R/RO - 0.04 - 1.5 - 10 \log d / dO - \alpha \]  
(Eq. 1)

or

\[ SL + 10 \log n - 15 \log RO - 10 \log R / RO - 0.04 - 1.5 - 10 \log d / dO - \alpha \]  
(Eq. 2)

where \( RO \) is the range in kilometres before transition to cylindrical spreading, \( R \) is the range between the source and the receiver in kilometres, \( d \) is the depth at the receiver in metres, \( dO \) is the depth at the sound source in metres and \( n \) is the number of sources. Depending on the seafloor, the minimum frequency that can propagate has a wavelength of four times the minimum water depth. However, a simpler cylindrical spreading model (Richardson et al. 1995) which also incorporates a constant based on the current study’s field measurements reflects the sound field from the single monopole source relatively well in the shallow water encountered in this study and is given by:

\[ SL - 10 \log (R) - 1.5 - \alpha \]  
(Eq. 3)

The spreading of turbine sound can be modelled well from these equations and are based on our field data; albeit equation 3 is slightly less conservative than equations 1 and 2 (Figure 5.3.5). It is important to note the circumstantial nature of the data and subsequent equations presented in the current study. Care needs to be taken when comparing to other locations and seasons because of the complexity in shallow water propagation due to sea floor characteristics, temperature, salinity and depths. These equations are also limited in predicting the sound field created by the potential cancellations of frequencies from several sources as this has been marginally controlled for in Eq 1, 2 and 3 by calculating the frequency dependant \( \alpha \). However, it is recommended in shallow coastal waters with largely homogenous seafloors, consisting of sand or mud, that these equations are used by regulatory agencies instead of simple geometric models for assessing the potential spatial scale of ecological impacts of underwater sound from anthropogenic sources, such as tidal turbines. The use of Eq. 1 and 2 in conjunction with accurate high order modelling is also recommended. The results of this study indicate that the continued use of simple geometric models of the propagation of underwater sound from such sources are likely to greatly
underestimate the spatial extent of the ecological impacts from these developments in the coastal environment and should not be used.
Chapter Six

General Discussion

6.1 Background

There is a need to identify and better understand the impacts of anthropogenic sound on the marine environment, especially estuarine environments, so that these impacts can be more effectively managed. This was the key motivation for the research reported in this thesis which aimed to investigate potentially significant anthropogenic sound sources and their influence on the settlement stage larvae of estuarine crabs. Estuarine crabs were selected for this study because of their robust nature and their known acoustic sensitivity which can be easily measured through behavioural assays (Popper et al. 2001; Lovell et al. 2005; Lovell et al. 2006). The main research aim of the thesis was composed of four goals as follows: (1) To describe a range of potentially significant anthropogenic underwater sounds in New Zealand; (2) To determine the settlement behavioural response of estuarine crab megalopae to natural estuarine sound and anthropogenic sound; (3) To characterise spatial and temporal patterns in ambient sound from two distinct estuarine habitats, and; (4) To investigate the propagation of tidal turbine sound from both single and twin monopole sound sources and evaluate current models of sound propagation in shallow coastal waters.

6.2 Underwater sound pollution in New Zealand

Sources of underwater anthropogenic sound in New Zealand are extensive and include sounds generated by vessels, industrial activities, marine construction, seismic surveying and petroleum exploration. Many of these sounds have been studied internationally, however, there are no published comprehensive studies of anthropogenic underwater sound sources in busy harbours, such as Auckland’s Waitemata and surrounding Hauraki Gulf. Research described in Chapter 2 not only filled this gap in the literature, but went further by investigating potential detection distances for generic fish and crustacean species and thus
estimated the potential risk of acoustic masking for these animal groups. All identified anthropogenic underwater sound sources in the Hauraki Gulf, except for the sound emitted from a motorway causeway, were of sufficient intensities and in the appropriate range of frequencies to potentially mask ambient sound levels over a considerable distance (several kilometres). Furthermore, the possibility of these sounds masking acoustic communication signals among fishes or crustaceans appears likely.

The finding in Chapter 2 that underwater sound produced by a range of vessels underway at varying speeds (11 – 46 km h⁻¹) was of high intensity (broadband levels > 158 dB re 1 µPa @ 1 m) within the Hauraki Gulf was consistent with previous overseas studies that have found that motorized shipping has greatly contributed to the increased ambient sound levels below 100 Hz in the open ocean by around 15 dB since the 1960’s (Ross 1993; Mazzuca 2001; Andrew et al. 2002). While the research in Chapter 2 investigated the acoustic characteristics of vessel sound, it did not examine in detail the extent to which vessel sound may already be contributing to increased ambient sound levels in the marine environment in New Zealand. This is an important consideration that warrants further research.

The issue of underwater sound pollution is becoming more topical following many recent publications concerning the potential impacts of anthropogenic underwater sound (Slabbekoorn et al. 2010). The Hauraki Gulf Marine Park, near Auckland, has a natural marine environment which attracts thousands of visitors annually and has been a focal point for human activity for nearly 1000 years (see auckland.co.nz). It is promoted as a unique place with distinctive landscapes and impressive biological diversity. However, ambient underwater sound levels within the Gulf are likely to be increasing due to growing numbers of pleasure craft, ferries, as well as greater numbers of freight vessels using the major shipping route to and from Auckland. One possible significant impact of vessel sound is the collisions between container ships and whales in the Hauraki Gulf, that are potentially the result of the soniferous whales becoming disorientated or confused by the vessel noise and exposing them to a greater likelihood of collision (Nowacek et al. 2004; Gerstein et al. 2005; Allen et al. 2012). Between 1989 and 2007, the carcasses of 38 Bryde’s whales have been reported in northern New Zealand and of those, 13 whales have been confirmed, or suspected, to have been killed by ship strikes (Behrens 2008). The potential link between vessel sound and whale fatalities due to ship collisions is a concern within the Hauraki Gulf,
and similar impacts appear to be likely in other heavily populated harbours around the world where ship strikes are already common (Douglas et al. 2008; Vanderlaan et al. 2009; Wiley et al. 2011; Mussoline et al. 2012; Parks et al. 2012). Research similar to this, and that described in Chapter 2, enables regulatory agencies to begin to quantify sources of anthropogenic underwater sound and manage their potential impacts (Pine et al. 2013).

6.3 A potential impact from underwater sound pollution in New Zealand: the influence of underwater turbine sound on crab larvae

The larvae of some crustaceans remain within coastal environments while other species venture tens of kilometres from the coast before making their way back to the nearshore before metamorphosing into juveniles (Paula et al. 2001; Radford et al. 2007; Daly & Konar 2008; Radford et al. 2008). The spatial distribution of reptant stage brachyurans may not be random but rather somewhat the result of directed movement by swimming megalopae responding to various settlement cues and distinguishing among potential settlement habitats. For example, the duration of the megalopal stage of *Callinectes sapidus* was found to decrease as temperature was increased from 15 to 30° C and optimum conditions for larval development between 15 – 35 psu and 24 – 30° C (Costlow 1967; Forward et al. 2001). Chemical cues associated with estuaries, such as humic acid, have also been found to mediate larval development (Forward et al. 2001). For example, the duration of the megalopal stage of *C. sapidus* decreased when exposed to estuarine water, even when diluted by 10% with offshore seawater (Forward et al. 1996).

Further evidence for active habitat selection in estuarine crab larvae is presented in Chapter 3, as estuarine sound was shown to consistently reduce the TTM in *Austrohelice crassa* and *Hemigrapsus crenulatus* (by 21 – 31 % compared to control treatments) when previously, reef sound has been found to have no effect of the TTM in *A. crassa*. This finding motivated the remaining research in Chapter 3 which identified a potential impact of turbine sound on the settlement of the megalopae of two estuarine crab species. Furthermore, the findings indicated that the frequency of turbine sound was more relevant in explaining the response of larvae rather than the sound intensity per se. This was significant because it suggested that other anthropogenic sound sources of similar frequencies and intensities to tidal and offshore wind turbines may also have an impact on the metamorphosis of these crabs. Preliminary
experimental results, not presented in this thesis, strongly suggested that the underwater sound from both a container vessel and a commuter ferry also interfered with the metamorphosis of megalopae of *A. crassa* (Pine, unpublished data).

The research in Chapter 3 established a potentially disadvantageous behavioural response by the megalopae of an ecologically important estuarine crab species to tidal turbine sound. Consequently, further research is strongly recommended to determine possible impacts on a wider range of species including commercially important fishes. The two estuarine species, *A. crassa* and *H. crenulatus* are extremely abundant within estuarine habitats, such as the Kaipara Harbour (i.e. over 550 m² for *A. crassa* (Sivaguru 2000)), and are important bioturbators and nutrient cyclers in shallow waters (Sivaguru 2000), as well as a key food source for juveniles of a range of commercial fish species (Paul 2000). Consequently, reduced recruitment and subsequently smaller populations of these crabs may have ecological effects which could also potentially impact local fish stocks.

A likely impact from both sea-based wind turbine and underwater tidal turbine sound is the masking of naturally occurring acoustic settlement cues in both larval crustaceans and possibly fishes (Slabbekoorn et al. 2010). Besides acting as a settlement cue, natural sources of underwater sound from suitable settlement habitats also have a strong influence on the swimming behaviour in crab megalopae of some species, with crabs orienting their swimming toward the sound source, presumably to assist them locating suitable settlement habitats (Radford et al. 2007). Although not investigated in the current research, it would seem likely that underwater turbine sound may also interfere with the orientation behaviour of swimming crab larvae, in the same way turbine sound has been shown to influence their settlement behaviour. By determining if this directional swimming response in crab larvae can be elicited by tidal turbine sound through future research, it will greatly assist in defining the extent that turbine sound may impact marine larvae. Further research might also examine the spatial distributions of juvenile crabs in relation to those of operating marine turbines.

### 6.4 Characterising ambient acoustics from potential tidal turbine sites

The masking of natural acoustic signals by underwater anthropogenic sound can only begin to be assessed when the characteristics of the ambient underwater sound is known (Lloyd et
The ambient acoustics of estuarine habitats is poorly described in the literature, with very few published studies (Bassett et al. 2010). This is despite the importance of this knowledge for understanding the possible role of ambient underwater sound in estuarine habitats to act as an orientation and/or settlement cue for larvae, as well as juveniles and adults of a range of species (Montgomery et al. 2006; Stanley et al. 2010). The results of the research presented in Chapter 4 showed significant temporal and spatial variation in ambient underwater sound within the Kaipara Harbour and a rise in spectral power during dawn and dusk periods in the 2 – 4 kHz bandwidth within the seagrass-dominated habitat and 6 – 9 kHz bandwidth in the mudflat habitat.

Previous studies concerning the propagation and modelling of reef sound have found ambient reef noise to propagate well beyond a reef (Radford et al. 2011). It has been shown that sound from a finite reef size propagates with virtually no attenuation for a distance offshore equivalent to the overall length of the reef, after which the sound level decreases as per cylindrical spreading and any seafloor attenuation (Radford et al. 2011). Consequently, the extent of reef sound as an orientation cue could be much greater than initially hypothesised (Radford et al. 2011). Recordings from the current study have shown that ambient sound emanating from estuarine habitats have similar intensities to reef sound, albeit with different spectral composition due to the absence of grazing echinoderms and soniferous reef fishes in these habitats. Therefore, it is likely that sound from estuarine habitats, such as those recorded during this study in the Kaipara Harbour, will also propagate over long distances well beyond the extent of the habitat. This would enable habitat-specific estuarine sound to be used as a long distance orientation cue by pelagic larvae, as well as mobile juvenile and adult fishes (Radford et al. 2011).

Research described in this thesis and previous studies have shown consistent behavioural responses of a range of marine organisms to underwater sound, many of which have the potential to be important in determining the spatial distribution and abundance of populations, such as through their influence on larval recruitment. How exactly the masking of natural sounds impact ecosystems in sites like the Kaipara Harbour is unknown at this time, however, the extent of the masking will depend on both the natural sound environment and the anthropogenic sound source. Given that the underwater sound produced by tidal turbines is of high intensity compared to the ambient underwater sound typically encountered
in estuarine environments, it is likely that the active frequencies of turbine sound has the potential to interfere with the metamorphosis of crab megalopae over a considerable radius around a turbine.

6.5 The significance of the results for managing underwater anthropogenic sound

In New Zealand, the anthropogenic sound impacts from any marine development must be adequately assessed under the Resource Management Act 1991. Understanding the propagation of any anthropogenic sound is critical in determining the impact zone (Erbe 2012; Erbe et al. 2012) and estimating the likely detection distances for fishes and crustaceans. Inadequacies in the understanding of the propagation of underwater sound in shallow coastal waters may result in underestimates of the propagation of turbine sound and therefore the potential extent of the areas impacted with anthropogenic sound. The findings in Chapter 5 illustrated this further by revealing marked differences in the sound fields generated by single and twin monopole sound sources; primarily a cumulative effect on sound levels from an increasing number of sound sources.

Complex modelling of sound propagation in shallow water coupled with field data has been shown to be relatively accurate under near ideal conditions with consistent depth changes, small surface waves and a homogenous seabed (Tindle & Deane 2005). However, in a practical sense, these models are difficult for resource management practitioners as they depend on an advanced understanding of physics which is not adequately accessible to regulatory bodies. Consequently, simple geometric spreading models are sometimes used to estimate sound impact zones in some cases. However, geometric spreading models are not precise in practice as they are based on two idealised scenarios (i.e., water of constant depth from the source and water of infinite depth from the source) within the range of which the actual values are thought to be reliably captured (Urick 1983; Richardson et al. 1995; Lurton 2002). Furthermore, these models lack the incorporation of further variation due to environmental factors, such as varying attenuation coefficients of seawater and the seafloor. The inaccuracy of geometric spreading models is illustrated by field data obtained in this thesis and is most likely due to their inability to incorporate the environmental variables that influence underwater sound propagation into the models. Consequently, the use of equations
presented in Chapter Five is recommended as a rapid preliminary assessment instead of simple geometric models.

The proposed new equations for estimating underwater sound spreading in shallow coastal waters presented in Chapter 5 are a much better fit to the field data at the time and location, however, the assumptions encompassing these equations must be recognised before attempting to apply them to other similar situations. One of the key assumptions is that the propagation medium is homogeneous in temperature and salinity which is important when dealing with anthropogenic sound propagation in, or around, estuarine habitats (Priestley & Thain 2010) where acoustic channelling may occur within haloclines (Urick 1983).

Another important consideration for determining the impact of anthropogenic sound on the marine environment is that the successful propagation of sound underwater depends on the depth of the water column (Richardson et al. 1995). The attenuation of any underwater sound at any given frequency is dependent on the depth of the water (Lurton 2002). This becomes important when attempting to manage potential anthropogenic sound sources in shallow bays (< 10 m). The development of the propagation equations presented in this thesis were done so at depths ranging between 9 and 36 m and were either compared against known anthropogenic sound sources from published studies, or actual field measurements. The similarities in both source levels and detection distances between the current and past studies supported the equations developed within this thesis.

Throughout this thesis, anthropogenic sounds of concern were defined as those within the audible frequencies of the receiver and of sufficient energy levels to overpower ambient sound levels (Slabbe Koorn et al. 2010). This definition relies heavily on the determination of the hearing thresholds of fish and invertebrates. Hearing sensitivity and thresholds vary among species and as a result, the stated detection distances for a generic fish or crustacean for the purposes of the research presented in this thesis requires a degree of caution. Hearing thresholds within fishes have been well studied and are generally lower (~ 50, 100 and 103 dB re 1 µPa in Carassius auratus, Astronotus ocellatus and Alosa sapidosomma, respectively (Kenyon et al. 1998; Mann et al. 2001)) for frequencies below 1 kHz (Kenyon et al. 1998; Mann et al. 2001; Egner & Mann 2005; Maruska et al. 2012). However, some fishes are capable of detecting ultrasonic sounds, with recorded auditory brain responses (ABR) as high
as 180 kHz in the American shad, *Alosa sapidosomma* (Mann et al. 2001; Higgs et al. 2004). Less understood are the hearing thresholds in invertebrates with only a handful of rigorous studies available (Popper et al. 2001; Lovell et al. 2005; Hu et al. 2009). Some crustaceans have, however, been found to be acoustically sensitive to low frequency sound (Lovell et al. 2005). While specific hearing thresholds of decapod crustaceans are largely unknown, the ability to respond to underwater sound in decapod larvae is very apparent, with several species being shown clearly to be able to respond behaviourally and physiologically to underwater sound (Jeffs et al. 2005; Radford et al. 2007; Stanley et al. 2010; Simpson et al. 2011), although the specific frequencies they are responding to are unknown. Most of the underwater anthropogenic sound sources characterised in this thesis consisted of low frequencies (below 2 kHz) and sound levels that were greater than the hearing thresholds of many fishes for frequencies below 5 kHz. This primarily led to the assumption that the anthropogenic sound source was detectable by a generic fish or crustacean if the received level was greater than a conservative 6 dB above the ambient sound level.

### 6.6 Rethinking underwater sound pollution in New Zealand

Currently, resource consent applicants in New Zealand are required to consider sound pollution and adopt the best available practicable options to avoid excessive underwater sound levels (which are currently well not defined) under the New Zealand Resource Management Act 1991. While such legal controls have the capacity to better manage underwater sound pollution, it depends on planners, administrators, companies and consultants maintaining an adequate understanding of underwater sound propagation and its potential environmental impacts. Defining generically acceptable sound levels is extremely challenging, and arguably impractical, as every anthropogenic sound source has different sound levels, frequency composition and duration; all of which may vary through time and by location. Furthermore, the extent and scale of environmental impacts from sound pollution are not currently well known, making it even more difficult to define acceptable underwater sound levels. In contrast to those activities requiring resource consent approval, there are currently few legal controls on the underwater sound emitted by vessels operating in New Zealand waters, despite the potential for this sound to cause environmental impacts.
The results from this thesis could aid regulatory bodies in developing guidelines for the management of underwater sound. Such guidelines would need to include the assessment of three key parameters:

1. Broadband source level and sound pressure levels of specific frequencies;
2. Dominant frequencies;
3. Duration and dynamics of the sound in question.

These three parameters are useful in determining the total extent of area that the underwater sound will affect, and most importantly the range of underwater habitats that may be impacted. Of most concern are likely to be habitats of high diversity and high productivity in shallow waters, such as reefs and shellfish beds, where a high diversity of acoustically active organisms are known to exist (Morton 2004).

Research described in this thesis, coupled with previous studies, has helped to promote greater understanding for the need to better manage anthropogenic sources of underwater sound in New Zealand. Resource management agencies in Auckland are now attempting to better manage anthropogenic sources of underwater sound under a new regional management plan which is currently being formulated (http://shapeauckland.co.nz/). Under this proposed new plan, underwater sound from any coastal development activity must be measured (and compared to a stated maximum decibel level), potential impacts assessed, and management contingencies put in place. Such contingencies range from ceasing sound emitting activity during marine mammal breeding seasons to requiring mechanical adjustments to equipment in order to reduce underwater sound emissions.

6.7 Conclusions and future research

This thesis investigated the acoustic characteristics of a range of sources of anthropogenic underwater sound in New Zealand’s coastal waters, and tested the potential for the sound produced by tidal and wind turbines to influence the settlement behaviour of estuarine crab species. Collectively, the results show underwater anthropogenic sounds of low frequencies (< 5 kHz), such as those produced by vessels and wind and tidal turbines, can travel
significant distances underwater, potentially mask important natural acoustic settlement cues, and can delay the metamorphosis of the megalopae of some estuarine crabs.

This research presented in this thesis extends our understanding of anthropogenic sound and the potential impacts on the marine environment. Several avenues for follow-on research were identified during the course of the research, including:

1. Advancing our understanding of anthropogenic sound sources in harbours

Research described in Chapter 2 found underwater anthropogenic sound may propagate great distances, traverse many habitat boundaries, and suggests the potential for it to mask important acoustic communication in some fishes. These findings warrant further research into the extent to which specific anthropogenic sound sources may already be contributing to ambient sound levels within harbours by comparing ambient levels during times when maximum and minimal anthropogenic sound sources are operating, or by comparing among a number of harbours where varying levels of anthropogenic sound sources are in operation. Long term acoustic monitoring of harbours will also provide new insights into trends in ambient sound levels over time.

2. Determining impacts of underwater turbine sound on the behaviour of crustacean and fish larvae

Research described in Chapter 3 found turbine sound significantly impacted the settlement and metamorphosis behaviour of estuarine crab megalopae within a laboratory experiments. Further research confirming whether turbine sound will interfere with metamorphosis when combined with natural underwater sound in situ, and whether the orientation responses of swimming crab megalopae are also affected by underwater turbine sound, is warranted. Such aims could be tested using choice chambers similar to the design developed by Irisson et al. (2009). Testing the orientation and settlement response of estuarine fish larvae to underwater turbine sound using field-based binary choice chambers and behavioural assays also has potential given the known importance of estuaries as nursery grounds for commercially important fishes.
3. Defining which specific frequencies of underwater turbine sound interfere with the metamorphosis of crab megalopae

It would be useful to define the specific frequencies of underwater sound from turbines that interfere with the metamorphosis of crab megalopae, as it may provide a route to manage the effects of underwater sound by adjusting the mechanics of turbines to alter the characteristics of their sound emissions. This may be tested by rearing megalopae under different known frequencies and measuring metamorphosis rates in a similar manner to Stanley et al. (2010, 2011) and Pine et al. (2012).

4. Determining which specific organisms or groups of organisms produce specific frequencies of natural estuarine sound

Research described in Chapter 4 found significant spatial variation in the ambient sound associated with habitats within the largest estuary in New Zealand, the Kaipara Harbour. The findings warrant further research concerning the identification of animals, responsible for the production of sound below 4 kHz, where most of the ambient sound energy was found. Such research could form the basis of a conservation management tool whereby habitat quality could be remotely assessed based on the specific frequency contribution to ambient acoustics.

5. Determining the long-term ecological impacts of anthropogenic sound

The current research shows that anthropogenic sound can significantly delay metamorphosis of brachyuran crab megalopae beyond an assumed temporal threshold and previous studies have suggested such behaviour may result in post-metamorphosis mortality and/or growth impairment in juvenile crabs. Consequently, long-term ecological effects could be prevalent in habitats within the acoustic range of persistent anthropogenic underwater sound sources, such as tidal turbines. Further research is needed to investigate possible differences in the development of juveniles following metamorphosis to establish if they result in any long-term ecological impacts.

6. Determining the hearing thresholds of a wider range of fishes and crustaceans in order to more accurately estimate sound propagation and potential impact zones
It was recommended in Chapter 5 that a newly modified equation based on field measurements (from both single and twin monopole sound sources) be used in place of traditionally employed spherical or cylindrical spreading models. Using the modified sonar equations in this thesis, it is possible to estimate detection distances for a generic fish or crustacean species. However, in order to improve the accuracy of these sonar equations, the hearing thresholds (i.e. detection thresholds) need to be better defined for a wider range of species over a wider range of developmental stages.


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