



Ecosystem approaches to management of pāua fisheries: Review and considerations

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Context

The Pāua Industry Council is in the process of reviewing how an “ecosystem approach” can be applied to management within the pāua industry. A recent advice paper reviewing the application of ecosystem based management to the pāua industry made a series of recommendations for how such an approach could be implemented in pāua fisheries (Fathom Consulting, 2018). The purpose of this literature review is to fulfil one of the recommendations of the advice paper, to:

“Commission a scientific overview of all relevant literature about the role of pāua in ecosystems and the environmental factors (natural and human-induced) affecting pāua distribution, abundance and behaviour. The purpose of this review is to:

- Identify knowledge gaps; and
- Inform fisheries plans and other management considerations.”

1. Pāua characteristics, life-cycle and key ecosystem interactions

1.1. Species and distribution

New Zealand’s black foot pāua (*Haliotis iris*) is a member of the abalone family, which are marine gastropod molluscs of the genus *Haliotis*. The blackfoot pāua is one of three species of New Zealand’s endemic abalone, which also include the yellow-foot abalone, *H. australis*, and the white-foot abalone, *H. virginea* (Poore, 1972a). The blackfoot pāua is the main commercially fished species of abalone in New Zealand. Pāua are distributed in a discontinuous manner around the coast of both main islands of New Zealand, Stewart Island, Chatham Islands, and the Snares Islands (Sainsbury, 1982a). Genetically distinct pāua stocks have been identified in the North Island, South Island, and Chatham Islands (Will et al, 2011; Will et al, 2015).

1.2 Pāua life-cycle and critical habitats

Pāua can reach maturity at around 60mm in shell length, although substantial spawning events do not occur until approximately 80mm, and there is well-documented regional variability in the length at maturity (Poore, 1972d, Naylor 2006). At maturity, sexes can be determined by the colour of the gonad, which is cream in males and green in females. Gamete production increases linearly with weight (McShane and Naylor, 1995), with large adult female producing as many as 11 million eggs (Poore, 1972d).

Pāua are broadcast spawners, meaning that mature adults synchronously release sperm and eggs into the water column (Hooker and Creese, 1995). With broadcast spawning, fertilization success depends on proximity of mature adults of both sexes. In *H. laevigata* recruitment failures from lack of fertilisation have been noted when densities fall below 0.3 animals m⁻² (Babcock and Keesing, 1999). Anecdotally, the largest spawning events are thought to occur around August (Radon, pers. comm. 2012), although spawning has also been observed between late summer and autumn in southern populations, with variability exhibited between localities and years. Spawning events are thought to be triggered by environmental cues such as storm events (Poore, 1972d).

After spawning, fertilized eggs are negatively buoyant for twelve hours until development into the trocophore stage then veliger stages (Tong and Moss, 1992; McShane, 1992; Stephens et al, 2006). At both these stages, larvae can actively swim for four to eight hours at a speed of 1-3mms⁻¹. Between days five and nine, larvae become competent. At which point they can settle, although they retain the ability to swim should an unsuitable substrate be initially settled upon (Tong and Moss, 1992). Crustose coralline algae are the favoured habitat for about eight weeks before settled juveniles seek out cryptic boulder habitats, where the diet is comprised of benthic diatoms (Kawamura et al, 1998). Variation in post settlement survival has been attributed to sedimentation, depth, predation, wave exposure, and grazing by adult pāua (McShane and Naylor, 1995). Due to the relatively short larval phase, dispersal of pāua larvae is reliant on the point of release, coastal topography and local hydrodynamic conditions (Stephens et al, 2006). Dispersal has been predicted using modelling approaches to occur regularly up to 200m from parents with a maximum dispersal of only 4km in calm conditions with normal current flows, although with tidal current flows dispersal can be extended to up to 50km with a maximum of 80km. While such conditions promote dispersal, they are less favourable for successful settlement, meaning repopulation of depleted areas is highly dependent on the right combination on hydrodynamic and habitat-related factors (Stephens et al, 2006). Despite the potential for long-range dispersal, it is generally thought that it is limited to 100s of meters (Naylor, pers comm, 2019).

Recently settled paua (<20mm) inhabit boulder or cobble field habitats in 0-5m depth, before moving to the shallow sub-tidal zone, usually in 0-2m. At this stage, pāua feed by grazing on crustose coralline algae and benthic diatoms (Aguirre and McNaught, 2010). Pāua generally emerge from cryptic habitats at the onset of maturity (approximately 80mm). Sexually mature pāua typically live in depths of 0.5 to 7m, although they may be found as deep as 15m, and occasionally in the intertidal zone (Poore, 1972b). Pāua are largely sedentary in nature (Schiel and Breen, 1991), and adults live in characteristic aggregations in rocky crevice and boulder habitats (McShane, 1995).

Pāua can grow to a maximum size of 200mm and growth has been shown to exhibit great regional variability both between fisheries management areas (Sainsbury, 1982a; Naylor et al 2006), and to scales of as little as 200m (McShane and Naylor, 1995). Growth is significantly influenced by sea surface temperature (Naylor et al, 2006), as well as wave exposure and habitat, which are correlated with food availability (McShane and Naylor, 1995). Factors affecting growth can also affect the size at sexual maturity, for example, decreased size at maturity is associated with warmer sea surface temperatures (Naylor et al, 2006). Growth rates are reduced after the onset of maturity (Hooker et al, 1997).

1.3 Food sources

Post-settlement, juvenile pāua feed by grazing on benthic diatoms and crustose coralline algae (Kawamura et al, 1998). As pāua reach maturity (about 80mm) pāua tend to feed more on seaweeds. Pāua can feed by grazing benthic algae, sedentary capture of passing drift algae, or by browsing substrate for algal matter (Shepherd 1973, Miner et al. 2006). Adult pāua feed mainly on drifting seaweed fragments (Allen et al, 2006). Feeding on attached algae is rarely reported, although it has been observed with high paua densities. It has been shown that high water movement (e.g., wave exposure) restricts active grazing and promotes drift feeding (Cornwall et al, 2009). Primary diet varies between locality depending on the floral composition (Allen et al, 2006). Preference has been shown for brown algae over reds and greens during laboratory feeding experiments and when whole plants are available, preference is shown for those that are most accessible (Cornwall et al, 2009).

1.4 Predators

A variety of taxa have been documented to prey on abalone including molluscs (whelks and octopus), starfish, crustaceans (crabs and lobster) and fishes (wrasse) (Jenkins, 2004). Starfish have been identified as the most significant predator of abalone in several studies (Aguirre and McNaught, 2013), and the seven arm starfish *Astrostele scabra* is commonly documented as the most significant predator of pāua in New Zealand (McShane and Naylor, 1997; Aguirre

and McNaught, 2013). The effect of predation by *A. scabra* is likely to be most significant during juvenile life stages of abalone. Negative relationships have been observed between the abundance of juvenile pāua (<10mm) and starfish *A. scabra* and *C. calamaria*. This correlation at juvenile stages suggests there is a strong influence of starfish on abalone populations at larger scales, making it a potentially important driver of pāua demography (Aguirre and McNaught, 2013). This has been observed in *H. rubra* where an increase in predation in a marine reserve was correlated to increase size in of emergent abalone, indicating the potential for predators to influence size structure of abalone populations (Pederson et al, 2008).

In pāua, both habitat complexity and algal complexity have been noted to play an important role in rates of predation by *A. scabra*. Predation is lowest in habitats with higher complexity, where the likelihood of an encounter between predator and prey is reduced. This relationship is most pronounced in juveniles, which illustrates the importance of cryptic juvenile habitats for predator avoidance. Predation may also be most significant at times of emergence and during transitions from algal to barren habitats (Aguirre and McNaught, 2013).

Newly settled pāua larvae can also be preyed upon by shell boring polychaetes. Increases in larval survival in complex coralline algae habitats may therefore be offset by predation by polychaete worms (McShane and Naylor, 1997). Mortality is also reported to be caused by wave exposure and pedal smothering by conspecific adults. Survival of recently settled larval has been reported to be higher in the absence of adults (Naylor and McShane, 2001).

1.5 Competition and interactions

The most commonly acknowledged competitor of abalone are sea urchins (Andrew and MacDiarmid, 1999; Jenkins, 2004; Strain and Johnson, 2009). In New Zealand this is the urchin *Evichinus chloroticus*. Potential competition exists between abalone and urchins as they both feed on drift seaweeds and share similar habitats. While they can co-exist, abalone can become locally extinct in areas of high urchin density (Naylor and Gerring, 2003), which has been explained by macroalgal depletion causing starvation in the abalone (Andrew and MacDiarmid, 1999).

Andrew and MacDiarmid (1999) examined relationship between *H. iris* and *E. chloroticus* in Fiordland. There was no evidence to suggest that interactions are of significant magnitude to limit populations of kina or that kina limit pāua populations. In other unpublished studies, negative relationships between pāua and kina at a range of spatial scales suggest that competition between pāua and kina is not a strong structuring process (Andrew and MacDiarmid, 1999). It has been suggested that urchins may out compete *H. rubra* when food is limiting, as

abalone tend not to inhabit urchin barrens. But when food is not limiting, abalone may be dominant as they can out compete urchins for space (e.g., in crevice habitats) (Jenkins, 2004). The same relationships are not exhibited in pāua and kina as food limiting barrens do not exist to the same extent as they might with other urchin species (Andrew and MacDiarmid, 1999).

In some instances (documented in California, Japan and South Africa) urchins have been shown to provide a refuge for juvenile abalone (Tegner and Dayton, 2000; Rogers-Bennet and Pearse, 2001; Day and Branch, 2002; Tarr et al, 2010). This has not been observed with *H. iris* in New Zealand, most likely because kina aggregations are relatively uncommon in critical habitats for juvenile pāua (0-2m). However, Wing et al (2015) reported that pāua can displace kina from shallow water wave sheltered habitats in the absence of fishing pressure.

2. Effects of pāua fishing on ecosystems

2.1 The pāua fishery

In New Zealand, approximately 700 t of wild pāua is commercially harvested each year across seven main quota management areas (QMAs), where commercial harvest ranges from 55 t to 195 t across QMAs. Most QMAs support significant but unquantified levels of recreational and customary fishing. Pāua are fished at a minimum legal shell length of 125mm, although pāua are usually harvested at larger sizes across the commercial fishery in recognition of variability in length at maturity and growth rates. Most pāua QMAs are estimated to have a biomass trending towards or at a level of 40% of their virgin biomass (B_0). However the catch levels in some QMAs are currently under review. New Zealand has one of the last remaining wild commercial abalone fisheries in the world.

2.2 Ecosystem effects of pāua fishing

Generally, ecosystem effects of fishing can arise from the methods of fishing (i.e., effects on habitats of the harvesting process), and the functional roles that the target species plays in its ecosystem (i.e., its relationship with predator, prey and competitors) (Anderson et al, 2011).

2.2.1 Effects on habitat

Pāua are harvested by free-diving (with the exception of the Chatham Islands where UBA is allowed) and using blunt knife-like tools to prise the animal off the reef. While there may be minimal amounts of damage to habitats caused by the

tools, catch bags or anchors, overall impacts are likely to be negligible compared to other fishing methods such as dredging and trawling (Anderson et al, 2011). In areas where there is high and repetitive fishing pressure, cumulative effects on habitat may be worth controlling. This could be achieved through divers codes of conduct (Jenkins, 2004).

Pāua harvesting also has no related by-catch and discards of unwanted species. There is a low level of returning undersized (sub minimum legal size or sub minimum harvest size) pāua to the reef, and mortality associated with this is thought to be low (Gerring et al, 2003), especially when best practise for harvesting are followed by divers.

2.2.2 Effects on ecosystems

Abalone's main interaction with their ecosystem is through feeding, predation and competition (Jenkins, 2004). Detail on these specific interactions is provided above in section 1. Generally, the effects on ecosystems of abalone fishing are considered to be low. Abalone are not regarded as a 'keystone' species, meaning they do not significantly affect the environment when they are removed (fished) (Zeeman et al 2014).

With regard to feeding, abalone do not have significant effects on kelp assemblages, as they feed mainly on drift algae (Cornwall et al, 2009; Zeeman et al, 2014). This means they are unlikely to have a structurally important role in reef ecosystems due to their feeding activity, on large scales (Jenkins, 2004; Valentine et al 2010). There is limited evidence that abalone may feed on attached seaweed occasionally, so there may be some negligible or unknown effects on kelp assemblages (Jenkins, 2004).

With regard to predation, ecosystem effects are also likely to be low. Any ecosystem impacts will depend on the strength of the dependencies between species of interest. While mortality inflicted on pāua by its main predator *A. scabra* may be significant, removal of pāua from the ecosystem is unlikely to impact the starfish. *A. scabra* is a food generalist and scavenger with a diet of mainly molluscs and crustaceans belonging to more than 60 genera (Town, 1980). The effect of pāua fishing on *A. scabra* is likely to be minimal except in areas where pāua are the main food source. In this situation there may be also be secondary effects on other sources of prey for the starfish e.g., yellow foot pāua (*H. australis*).

Any significant ecosystem effects of abalone fishing are likely to be related to competition for space with other reef grazers (Jenkins, 2004). For pāua, the main competitor is kina (*E. chloroticus*) (Andrew and MacDiarmid, 1999). From

studies examining relationships between pāua and kina, there is limited evidence to suggest removal of pāua could significantly affect kina populations (Andrew and MacDiarmid, 1999). Although there may be instances where intense grazing on kelp by kina can create 'barrens' (Tegner and Dayton, 2000), this is unlikely to be sufficient to limit pāua productivity in most regions of New Zealand where pāua are abundant. While there is some anecdotal evidence that the establishment of kina barrens may also exclude pāua from large areas of available habitats, this has not been documented in areas studied to date (Andrew and MacDiarmid, 1999). However, this is a difficult relationship to study as pāua tend not to inhabit areas where kina form extensive barrens. There is some evidence that the presence of abalone may exclude other invertebrates settling (Valentine, 2010; Wing et al, 2015). The overall ecosystem effect of fishing on these relationships is likely to be low, unless fishing was sufficient to limit abalone numbers in these habitats.

2.2.3 Disease

Another notable ecosystem effects of fishing is the transmission of pests or diseases. While there are multiple examples of disease-related mortality events in other abalone species (e.g., Abalone viral ganglioneuritis (AVG) in *H. rubra*), there are no comparable cases in pāua. This risk of disease transmission in pāua is therefore considered to be low. As a precaution, the industry now has an established protocol for the reporting and testing of any cases of suspected disease.

2.3 Summary

In summary, the overall ecosystem effects of pāua harvesting are likely to be low. This is largely due to its preference for feeding on drift algae, and because it is only a small component of the prey of its major predators. Any ecosystem effects are likely to be related to competition for space with other reef grazers (specifically kina). Despite anecdotal evidence of kina barrens (and their well documented ecosystem effects), there is no evidence that removal of pāua from the ecosystem contributes to the processes.

3. Environmental factors (natural and human induced) affecting pāua distribution and behaviour

Any ecosystem effects must be considered in the context of a background of other environmental effects, which may alter the strength of ecosystem relationships (Jenkins, 2004). There are a variety of environmental factors that can influence pāua distribution and behaviour. Natural environmental factors

have been reviewed in detail in the first section, so are briefly summarised below. Human induced environmental factors can be broadly categorised as sedimentation and climate change (specifically ocean acidification and ocean warming), and can be specifically linked to natural factors affecting pāua.

3.1 Natural environmental factors

Available rocky reef habitat is the primary determinant of pāua distribution. Pāua are distributed around the extent of New Zealand's coastline but is discontinuous where habitat is disrupted by large areas of sand or sediment. On occasion, sand movement can cover available habitat, and there are several recent anecdotal reports of sand smothering pāua, or reducing the available habitat causing pāua to eventually be dislodged by swells. This is likely to be a relatively minor source of mortality but it may change as the frequency of storm events (potentially influencing sand movement) increases. Another obvious but isolated instance of loss of habitat was the 2016 Kaikoura earthquake. The uplift from the earthquake caused approximately 25% of the available habitat (by fished area) to be lost, which caused significant pāua mortality.

Sea temperature has a significant influence on the characteristics of pāua populations. Pāua are found in water ranging in temperature from about 9°C to 21°C, but temperature has a major effect on growth rate and maximum size, with faster growth rates generally being observed in cooler waters (Naylor et al, 2006). Temperature is the dominant driver of pāua growth, but water motion, food availability and habitat structure are also influential (McShane and Naylor, 1995; Naylor et al, 2006). Growth rates after maturity are significantly lower in areas where the maximum sea temperature is above 17°C. This means that sea temperature determines the regions where pāua are able to grow through to MLS (125mm), and explain why the vast majority of the commercial fishery exists from the Wairarapa south.

3.2 Human induced environmental factors

3.2.1 Sedimentation

Sedimentation is process of particles settling or being deposited as sediment, and is driven by geology, topography, active tectonics and rainfall (Jones et al, 2008). New Zealand's shallow coastal zone, proximity of large urban centres to coastal areas and, extensive areas of forestry and agriculture make sedimentation the most significant land-based stressor on our marine environment (Morrison et al, 2008). Sedimentation has been identified as a threat to all relevant pāua habitats and ecosystems including intertidal and sub-tidal reefs and kelp forests in sheltered and exposed costs (MacDiarmid et al,

2012). Sedimentation can have significant adverse effects on pāua at critical life stages, as well as macroalgae which are a critical component of pāua ecosystems.

3.2.1.1 Effects of sedimentation on pāua

Sedentary marine invertebrates such as abalone, are particularly vulnerable to the effects of sedimentation, because they require stable sub-strata for settlement and attachment (Onitsuka et al, 2008). Sedimentation has been observed to affect pāua survival and behaviour at all life-stages (McShane and Naylor, 1995; Kawamura et al, 1998; Chew et al, 2013; Sainsbury, 1982a).

Sedimentation has been identified as a significant factor influencing variability in recruitment success in pāua (McShane and Naylor, 1995). At the veliger (free-swimming) stage of pāua larval development, larvae settle preferentially on surfaces covered in crustose coralline algae (McShane, 1992; Kawamura et al, 1998). After settlement, veliger larvae start to metamorphose (turn into recognizable pāua), and graze on benthic diatoms (Kawamura et al, 1998). Layers of sediment on preferred settlement habitats will interfere with settlement cues, and decrease the amount of physical area available for settlement, as well as reducing surface area for grazing during metamorphosis. A study on *Haliotis diversicolor* found that the rate of larval metamorphosis was reduced by increased sediment, and that changes were dependent on both quantity and quality of sediment. They concluded that both settlement success and post settlement mortality were highly influenced by terrestrially derived sediments in wild abalone (Onitsuka et al, 2008). With pāua, sediment has been shown to induce mortality in settled larvae, with effects persisting even after sediment is removed. These effects are therefore likely to be manifested in the event of one-off heavy rainfall leading to erosion and short exposures to high sediment levels (Philips and Shima, 2006)

Juvenile pāua (<80mm) live in cryptic habitats, primarily under large boulders (Roberts et al, 2007). This requires crevices small enough to avoid major predators such as the seven arm starfish (*Astrostele scabra*), but large enough to allow the flow of drift seaweed for feeding. Chew et al (2013) tested the effects of sediment loading on juvenile pāua. While they did not find direct effects on mortality, behaviour was modified as pāua had to move out of cryptic habitat as sediment accumulated. This exposes juveniles to an increased risk of predation and potentially dislodgement (Chew et al, 2013).

High rates of sedimentation have been linked to direct mortality in adult pāua (Sainsbury, 1982a; Schiel, 1993; Preece, 1998). This has been explained by sudden localised changes in sediment distribution after storm swells, which forced pāua off preferred substrates or buries them (Sainsbury, 1982a). Adult pāua are largely sessile and require clear, flat substrate to live and feed.

Increased sedimentation essentially reduces the carrying capacity for adults by reducing good habitat, which can hinder the rate of population rebuild after mortality events (S. Stanley, pers. comm., 2014). There are numerous anecdotal reports of pāua mortality following sedimentation and storm events, most recently at Maungamanu north of Kaikoura (April, 2019).

3.2.1.2 Effects of sedimentation on macroalgae

As outlined in earlier sections, macroalgae are a critical element of determining ecosystems pāua can inhabit. While pāua are likely to be negatively influenced by sedimentation before most seaweed species, sedimentation has been documented to have effects on macroalgae at various life stages. One of the most studied macroalgae in relation to effects of sedimentation is the giant bladder kelp *Macrocystis pyrifera*. *M. pyrifera* can be found over large extent of the pāua distribution, and is particularly abundant in areas where pāua tend to grow the fastest, making it a critical component of pāua ecosystems, It is important to pāua primarily as a food source as drift, and also in buffering wave energy to protect juvenile habitats, and has a role in determining larval dispersal dynamics. *M. pyrifera* beds exhibit high variability in abundance over different time scales. Sedimentation has been identified as one of the most significant drivers of variability in *M. pyrifera* abundance (via storm and high rainfall events) (Pirker, 2002), although there is evidence that *M. pyrifera* can tolerate more sediment than other algal species (C. Cornwall, pers. comm, 2019).

The settlement and early development of microscopic stages of kelps can be particularly vulnerable to sediments, with both lethal and sub-lethal deleterious effects occurring through prevention of propagule settlement and attachment, smothering of already settled propagules, abrasion or complete removal of propagules and early life-history stages (Schiel et al, 2006), and the inhibition of photosynthetic activity via reductions in light levels limiting the recruitment of juveniles (Devinney and Vorse, 1978; Pirker, 2002; Geange et al, 2014).

Sedimentation can also limit the productivity of adult kelp by increasing water turbidity and reducing the availability of light for photosynthesis, decreasing productivity. The settlement of fine sediments on the laminae of adult and juvenile plants may also shade photosynthetic tissues and inhibit nutrient exchange required for growth (Pirker, 2002; Geange, pers comm., 2012). The thickness of the grain can be important as increasing coarseness allows greater light diffusion and enhanced growth in some weeds (Geange et al, 2014). The magnitude of the effects of sedimentation on all these life stages are influenced by the local hydrodynamics, and the grain size of the sediment particles (Schiel et al, 2006).

3.2.2 Climate change considerations

There is an increasing awareness of the effects of climate change on the productivity of fisheries. Two key climate change considerations relevant to pāua are ocean acidification and increasing sea surface temperature.

3.2.2.1 Ocean Acidification

Ocean acidification is the gradual decrease in pH (or increased acidity) of the ocean caused by increasing amounts of atmospheric carbon dioxide. Levels of ocean acidification are affected by temperature, being slightly increased in cooler waters, which are where the most productive pāua fisheries exist. Lowered oceanic pH can have adverse effects on invertebrates that are reliant on carbonate for shell production (such as pāua) (Cunningham et al, 2016).

Reduced pH has been shown to effect larval survival and development. This is thought to be caused by a reduced rate of shell deposition, meaning larvae are dependent on energy stores for longer in the veliger stage, thus increasing energetic requirements (Cummings et al, in review). Ocean acidification could also limit pāua recruitment, as it impairs growth in crustose coralline algae (Cornwall et al, 2013) thus decreasing suitable settlement habitats and food sources for larvae. While this hasn't been observed specifically in pāua, it has been shown to influence other species (e.g., corals in Webster et al, 2013). Ocean acidification has been shown to reduce growth rates in juveniles and while no direct effects have been observed on the survival of adults, it increases shell erosion which can affect growth, and potentially physical condition and respiration rates increasing overall energetic requirements (Cunningham, 2013).

The effects of long-term ocean acidification could therefore cause a reduction in pāua recruitment, by limiting larval development and juvenile growth, as well as affecting fishery dynamics by decreasing growth rates.

3.2.2.2 Changing sea surface temperatures

Temperature influences many aspects of the pāua life-cycle and behaviour, so the long-term trends of increasing sea temperatures are likely to have significant effects on pāua ecosystems.

3.2.2.2.1 Sea temperature and pāua

At early life stages, increased temperature has been shown to increase the speed of larval development in pāua (Tong et al, 1992), and juveniles have also been shown to grow faster at higher temperatures (Tong, 1982), however this relationship weakens as pāua grow larger (Searle et al, 2006). Growth rates in

adults are significantly influenced by temperature with growth, and maximum length being larger at cooler temperatures (Naylor et al, 2006). Length at maturity has also been shown to increase with cooler temperatures, although this is likely to be due to reduced growth rates after the juvenile phase in cooler water (Naylor et al, 2006). Temperature therefore plays an important role in controlling pāua population characteristics across through their range, e.g., pāua are typically only fished from the Wairarapa south where waters are cool enough to promote growth in pāua over 125mm regularly. In red abalone (*H. rufescens*) increasing sea temperature has been shown to restrict growth rates and reproduction (Vilchis et al, 2005). Changing sea surface temperatures could therefore potentially restrict the range of pāua, and in particular the range of the fishery where pāua grow over the minimum harvest size. Disease susceptibility in abalone is also linked to increased water temperatures (Vilchi et al, 2005; Rogers-Bennett et al, 2010), with disease outbreaks in pāua typically associated with water temperatures over 20°C (Diggles and Oliver, 2005).

3.2.2.2 Sea temperature and macroalgae

Sea temperature can also affect the abundance and distribution of macroalgae which are critical components in ecosystems pāua inhabit (as outlined earlier). Temperature plays an important role in the life cycle of *M. pyrifera*. For example during summer, kelp forest biomass declines in response to nutrient limitation, as nutrient exchange is inhibited by warmer temperatures (Geange, pers. comm., 2014, Ignacio Vilchis et al, 2005; Mabin et al, 2019). Temperature therefore has an important role in defining the extent of the range that *M. pyrifera* typically occupies (generally south of Castle Point on the east coast and Kapiti Island on the west coast of the North Island). Increasing temperatures could therefore also restrict the range of key macroalgal species such as, *M. pyrifera*, which could have downstream effects for pāua ecosystems and fisheries.

3.2.2.3 Marine heat waves

A related phenomenon to changing sea-surface temperature is marine heat waves. Marine heat waves have been defined as a prolonged discrete anomalously warm water event (Hobday et al, 2016). Marine heatwaves are occurring in environments that support paua ecosystems with increasing frequency and magnitude (Mundy, pers. comm. 2019). There have been anecdotal reports of paua mortality and starvation associated with marine heatwave events (e.g., on the Chatham Islands (PAU4) in 2019). However, it is difficult to isolate a marine heat wave as the main driver of mortality or starvation in the context of other processes such as sand movement due to storms, and variability in kelp abundances.

3.2.2.3 Storm events

There is evidence of increasing frequency of storm events with climate change (Easterling et al, 2000). Storm events drive different physical processes that have potential impacts on pāua ecosystems. Reproductive cues in pāua are poorly understood, but it is thought that storm events serve as a cue to induce spawning (Poore, 1972d), so increased frequency of storm events may alter the timing and magnitude of spawning events. After fertilization storm events may also affect the survival of larvae and influence the distance of dispersal (Stephens et al, 2006), so have the potential to influence population dynamics. Large swells associated with storms can also disrupt juvenile boulder habitats causing mortality at critical life stages and limiting recruitment to the fishery. Large swell have also been observed to dislodge and cause mortality in adult pāua, particularly when associated with high rainfall and sedimentation events. There are several documented cases of this occurring, most recently at Maungamanu north of Kaikoura (April, 2019). This has been explained by sudden localised changes in sediment distribution after storm swells, which forced pāua off preferred substrates or buries them (Sainsbury, 1982a).

Storm events can also drive the change in abundance of kelp forests especially when associated with high levels of sedimentation after rainfall events (Pirker, 2002). Storm events are associated with heavy rainfall, which is also a driver for increased sedimentation (Glade, 2003), potentially amplifying adverse effects on pāua populations and macroalgae described above under sedimentation.

3.2.3 Kelp Harvesting

The importance of kelp species to pāua ecosystems is discussed in earlier sections. Large-scale kelp harvesting could have adverse effects on pāua ecosystems primarily by limiting food supply. This would be unlikely to have a significant impact unless there was high intensity harvesting in areas of high pāua productivity. There is a TACC for attached *M. pyrifera* of 1,500 t around New Zealand, with 1,200t of this being allocated to the east coast of the South Island. While this is an important area for pāua fisheries, a minimal amount of the TACC for kelp is utilised, so current scales of kelp harvesting are unlikely to have an affect on pāua fisheries, unless large localised areas of kelp were denuded. Ecosystem effects on pāua fisheries should be considered if this TACC becomes utilised in concentrated areas.

4. Application

4.1 How can this understanding be used to inform management considerations?

4.1.1 Ecosystem effects of pāua fishing

While ecosystem effects of pāua fishing are likely to be low, there may be cumulative effects in areas of high fishing pressure, or where large-scale serial depletion has removed significant amounts of pāua from the ecosystem. These effects could be managed through harvester education and industry management initiatives.

- Harvester training:
 - Review and update training material to cover potential ecosystem effects of pāua fishing to create awareness with harvesters.
 - Update diver codes of conduct (through annual operating plans) to suggest ways to minimise impacts to pāua ecosystems, e.g., avoid anchoring near recognised recruitment habitats, protocols to report potential disease incidents and for minimising risks of disease transmission
 - Encourage harvesters to document and report any unusual ecosystem changes.
- Industry Management
 - Catch spreading to minimise risks of ecosystem effects caused by localised depletion of pāua, and cumulative affects of activity on ecosystems.

4.1.2 Environmental effects on pāua

There is evidence that environmental effects have considerable potential to impact pāua fisheries and their ecosystems. Environmental effects are variable and influenced by multiple drivers, making them hard to manage in isolation. The focus should be monitoring of environmental changes that impact pāua fisheries, and implementing management actions if changes are detected:

- Monitoring environmental changes:
 - Creating harvester awareness of environmental affects on pāua ecosystems by incorporating material in harvester training materials. Changes could be monitored through diver surveys to record observations and changes in ecosystems and stressors.
 - Coordination with research institutions monitoring sea temperature and ocean acidification (University of Otago), and habitat and ecosystem monitoring (E.g., Canterbury University intertidal monitoring in earthquake affected areas of PAU3 an PAU7).

- Monitoring the abundance of pāua at the life-stages most vulnerable to adverse environmental effects (e.g., monitoring recruitment using pāua ‘motels’).
- Monitoring marine heat waves and their effects on paua health.
- Management responses to environmental changes
 - Incorporation of projected changes into long-term management strategies and initiatives (e.g., to review minimum harvest size in areas with detected increase in sea surface temperature, translocation to areas of cooler currents, habitat enhancement and kelp restoration in areas of depletion).
 - Reviewing catch-spreading efforts in the event of marine heatwaves if mortality is observed.
 - Reduce cumulative effects of environmental stressors by managing individual local stressors.
 - Development of best practices to reduce sedimentation in near shore ecosystems (strengthened by the existence of Fisheries Plans?).

4.2 What are the knowledge gaps?

There has been a considerable amount of research to characterise the life-cycle and behaviour of pāua, as well as many ecosystem interactions. There is also a large body of research documenting the effects that various environmental factors can have on different life-stages and other key species (e.g., macroalgae). These studies are often very focused on specific life-stages and single effects. Outcomes often suggests there is need for concern with how pāua fisheries are managed in light of changing environmental stressors, however it remains impossible to quantify the total impact that these stressors are having on the fishery, making it difficult to react confidently with management decisions.

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