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Research Article

Detection of the tropical mussel species *Perna viridis* in temperate Western Australia: possible association between spawning and a marine heat pulse

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Abstract

In April 2011 a single individual of the invasive mussel *Perna viridis* was detected on a naval vessel while berthed in the temperate waters of Garden Island, Western Australia (WA). Further examination of this and a nearby vessel revealed a small founder population that had recently established inside one of the vessel's sea chests. Growth estimates indicated that average size mussels in the sea chest were between 37.1 and 71 days old. Back calculating an 'establishment date' from these ages placed an average sized animal's origins in the summer months of January 2011 to March 2011. This time period corresponded with an unusual heat pulse that occurred along the WA coastline resulting in coastal waters >3 ºC above normal. This evidence of a spawning event for a tropical species in temperate waters highlights the need to prepare for more incursions of this kind given predictions of climate change.

Key words: *Perna viridis*; spawning; climate change; invasive species; heat pulse

Introduction

Anthropogenically induced climate change and non-indigenous species introductions are regarded as two of the greatest threats to global biodiversity (Vitousek et al. 1997; Halpern et al. 2008) and have had a myriad of effects and impacts on the distribution and diversity of species (Vitousek et al. 1997; Ruiz et al. 2000). From habitat modification (Wallentinus and Nyberg 2007), ecosystem engineering (Crooks 2002), species displacement (Erickson 1971), to competition for resources (Usio et al. 2001; Vitousek et al. 1997) each of these factors is a threat. However, it is perhaps the synergistic potential of both these threats that poses the greatest risk to marine biodiversity.

In its native tropical waters of the Asia-Pacific and Indo-Pacific regions, the Asian Green Mussel, *Perna viridis* (Linnaeus, 1758) forms the basis for an important aquaculture industry (Qasim et al. 1977; Sivalingam 1977; Sreenivasan et al. 1989; Zhang et al. 1997; Chalermwat et al. 2003). However, the very characters that contribute to its success as an aquaculture species also contribute to its success as an invasive species and it is consequently one of the most commonly encountered invasive species detected on vessels entering Western Australian waters. *P. viridis* displays rapid growth and early onset of maturity, and is highly tolerant to a broad range of water temperatures (native range 11-32°C), salinities (18-33 psu, turbidity, and pollutants (Sivalingam 1977; Lee 1986; Benson et al. 2001; NIMPIS 2011). Additionally, broadcast dispersal of planktonic larvae, the capacity to settle on a variety of hard surfaces and an ability to survive in a wide range of depths (from the intertidal to 42m) allows *P. viridis* to outcompete existing native assemblages, resulting in changes to community structure and trophic relationships (NIMPIS 2011).

Perna viridis has a gregarious nature and is a strong spatial competitor with juvenile settlement densities reaching up to twelve thousand m^{-2} (Power et al. 2004), often resulting in populations exhibiting thick carpet-like growth. Rajagopal et al. (1991) highlight the fouling potential of this species and report that of the 570 tons of fouling lodged in the intake tunnels of a power station, *P. viridis* constituted approximately 72% (411 tons). Baker et al. (2002) report densities of *P. viridis* in intertidal areas of Tampa Bay between 3,675 and 4,117 individuals m^{-2}. It has been suggested that this species may eventually become the marine equivalent of the freshwater zebra mussel (*Dreissena polymorpha*) which, cost nearly $1 billion in management efforts alone, over a span of 15 years in invaded North American and European waterways (Power et al. 2004).

Perna viridis is currently not known to be established within Australia and due to the invasive characteristics detailed above, is listed as a high priority pest species under the 'Australian Government National System for the Prevention and Management of Marine Pest Incursions' (National System 2011). In April 2011, a single *P. viridis* approximately 60 mm long was detected by a navy clearance diver on the naval frigate *HMAS Anzac* moored at the Australian Naval base (*HMAS Stirling*) at Garden Island, Western Australia.

In early 2011 a significant warming event was occurring, with this warm patch covering an area extending from Ningaloo to the Capes region (over 1200 km) and to almost 200 km offshore (Pearce pers. comm). Warming of Australia's temperate coastal regions is predicted to increase mean water temperatures by up to 3°C by 2070 (Lough 2009). However, associated with mean climatic changes are extreme events like heat pulses or heat waves which may amplify and even accelerate adverse effects. Heat waves have been defined as a period of at least three to five days during which the mean or maximum temperatures are at least 3-5ºC above normal are observed (Meehl and Tebaldi 2004). These extreme climatic events are predicted to increase in both severity and frequency as consequence of global climate change (IPCC 2007a, b).

There is a growing body of evidence that suggests increasing temperature is correlated with the growth and success of invasive species. Sorte et al. (2010b) document that oceanic warming, along with increasing distribution vectors could have contributed to the spread of non-indigenous fouling species documented by others (Lambert and Lambert 1998) in California. Sorte et al. (2010a) predict that the invasive ascidian *Didemnum vexillum* in Bodega Harbour, California will start to dominate the natural fouling assemblages as temperatures increase. It has also been proposed that the spread of *Carcinus maenas* from Victoria to Tasmania (Australia) may have been facilitated by increasing water temperatures (Thresher et al. 2003). The detection of *P. viridis* in our coastal waters was cause for great concern and triggered an emergency marine pest response by the Western Australian Department of Fisheries (DoF), Defence Services Group (DSG) and the Australian Navy. *Perna viridis* has become a successful invader in many areas due to their tolerance to a wide range of temperatures (Rajagopal et al. 2006).

Materials and methods

The emergency response had three components: (1) a comprehensive in-water followed by a dry-dock inspection (under DoF guidance) of the *HMAS Anzac* and the adjacently moored vessel *HMAS Arunta*; (2) a delimiting survey of wharves and submerged infrastructure within *HMAS Stirling*; and (3) implementation of a longer-term monitoring program to verify the presence or absence of any established *Perna viridis* populations.

In-water delimiting survey of vessels

Navy clearance SCUBA divers undertook visual inspections for *Perna viridis*. Divers were briefed on what to look for prior to entering the water. A series of divers were deployed swimming approximately 2 m apart. Divers proceeded along the hull from stern to bow examining all areas of the vessel. Navy clearance divers did not detect any further *Perna viridis* during their searches of either HMAS *Anzac* or HMAS *Arunta*. However, divers did report significant fouling present on both vessels. The amount of fouling was sufficient that no degree of assurance could be made regarding the presence or absence of any more *Perna viridis*. As such, the Navy agreed to dry-dock both vessels for further examination. This agreement to dry-dock the vessels also coincided with a plan by the Navy to undertake maintenance on these vessels.

Dry-dock inspection of vessels

Vessels were cold-towed across Cockburn Sound and put into in dry-dock at Henderson, W.A. Both vessels were inspected by the author and a representative of the Navy.

While heavily fouled with another mytilid species, no *Perna viridis* were detected anywhere on the HMAS *Anzac.*

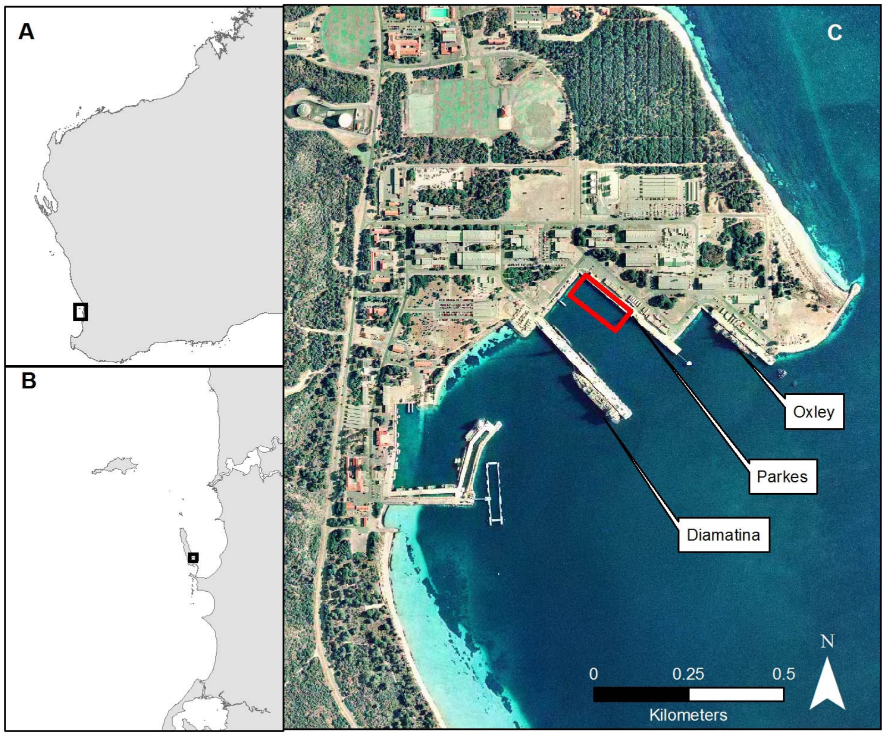

Figure 1. Map of Western Australia (**A**) showing location of Garden Island (**B**), (**C**) Location of the three wharfs (Oxley, Parkes and Diamantina) surveyed for the presence of *Perna viridis*. Note: infected vessels were berthed at Parkes wharf (denoted by red box).

The inspection of the HMAS *Arunta* revealed four *Perna viridis* from the external areas of the hull. The first was located on the external grate of a port-side sea chest, and the second on the port-side propeller housing. Two more animals were detected on the starboard propeller. An examination of internal sea chest and cavities revealed that most were clean of any biofouling. However, examination of one sea chest on the port side of the vessel (the same sea chest that the first mussel was collected from) revealed numerous juvenile *Perna viridis*. Inspectors collected all animals within reach for further examination (see results section).

In-water delimiting survey of infrastructure

As both *HMAS Anzac* and *HMAS Arunta* had remained stationary since entering Garden Island, a delimiting survey of the immediate surrounding area was implemented, targeting Parkes, Oxley and Diamantina wharves (Figure 1). An inspection of fouling communities present on the wharf pylons revealed a diverse, yet spatially consistent, fouling assemblage. Fouling communities exhibited vertical zonation with an upper margin of foliose green algae (such as *Ulva* sp.), followed by bands of barnacles, then mussels (at 1-3m) and a predominantly ascidian and sponge dominated assemblage below this. Following initial visual assessment, the decision was made to concentrate the sample effort at 1–3 m below waterline, where mussels occurred. This area corresponded to depths where populations of juvenile *P. viridis* have been recorded by other authors (Cheong and Chen 1980). Navy clearance SCUBA divers then undertook visual inspections for *Perna viridis*. Divers were briefed on what to look for prior to entering the water. Divers were also instructed to collect all mussel fouling from the pylons for examination on the surface. Divers scraped all material from the pylons into pre-labelled calico bags. Bags were relayed to the surface for examination (see Piola and McDonald 2012 for more details).

Long-term monitoring

The sampling methods used and the sorting of material collected from all accessible jetty areas was extensive and deemed rigorous. However, given the small size of specimens collected from the HMAS *Arunta*, it is possible that areas examined could have contained specimens that, at the time of survey, were too small to detect

Figure 2. Different colour morphs of adult (**A**) and juvenile (**B**) *Perna viridis*. Not all juveniles displayed markings similar to those illustrated, some displayed the bright green colouration more typical of this species (Scale bar in **A** is cm; scale in **B** is mm). Photographs by Justin I. McDonald.

visually. Thus, there was insufficient confidence to clearly deny the presence of *Perna viridis* at Garden Island. Further surveillance was conducted at 6 and then 9 months after initial detection. This time period was deemed suitable for detection of any animals that may have been too small to be detected in the initial survey, based on the age/growth rates determined below. This subsequent surveillance followed the same process as the initial delimiting surveys. The aim of this subsequent surveillance was to detect any *Perna viridis* that may have established from the spawning event and that had now grown to a size detectable by divers.

Results

In-water delimiting

In total, approximately one tonne of biofouling was examined by DoF staff from the wharf pylons at *HMAS Stirling* over a three day period, with no *P. viridis* being detected.

At six and nine months after the initial detection on *HMAS Anzac*, follow-up surveys were conducted of wharf infrastructure.

No further *P. viridis* were detected.

Demographics of Perna viridis detected on HMAS Arunta

A total of 201 *P. viridis* were collected from *HMAS Arunta* while in dry-dock. Four were collected from the external hull and a further 197 from the accessible areas of the sea-chest. Based on the size of the animals collected there appeared to be two cohorts, the first cohort comprised of mussels collected from external surfaces, which were much larger (range = 41.4-47.0 mm shell length; mean 45.03 ± 1.3 SE; n= 4) than those of the second (sea-chest cohort) (range = 2.8-17.19 mm shell length; mean 9.61 mm ± 0.20 SE; n=197). According to expert advice (Richard Willan, Northern Territory Museum), the first cohort were likely to have established while the vessel was in South East Asian waters, approximately six months previous.

Notably, many of the juveniles collected from the sea chest did not display the typical bright green colouration characteristic of this species; rather they had a dull olive green/brown colouration and zig-zagging bands of brown (Figure 2). This latter patterning is a documented, albeit less common, juvenile colouration for *P. viridis* (identifications verified by Richard Willan).

Estimating age

As the vessel had been in operation in both New Zealand and South East Asian waters (Singapore and Malaysia), there was early debate over whether the species was *P. viridis* or *P. canaliculus*. However after it was confirmed as *P. viridis*, the key management aim was to

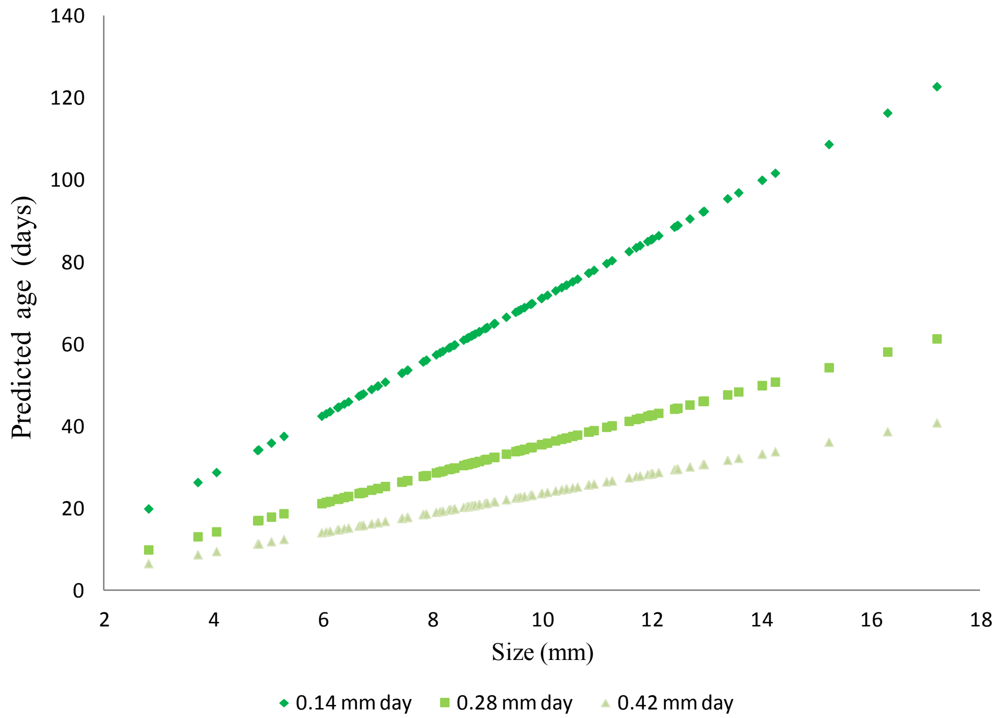

Figure 3. Predicted 'spawning date' of *Perna viridis* juveniles collected from sea chest of HMAS *Arunta*, based on published growth rates (Cheung 1993; Lee 1986; Narasimhan 1980; Rajagopal et al 1998; Rao et al 1975). Upper values (0.14 mm/day) = minimum growth rate, centre values (0.24 mm/day) = mean growth rate, lower values (0.32 mm/day) = maximum growth rate.

determine if the sea chest cohort had been missed in earlier inspections and were from elsewhere or if alternatively they had spawned in WA.

No records of *P. viridis* growth in cooler temperate waters were available to the author. As such, age estimates of *P. viridis* found at Garden Island were based on a range of growth rates from tropical or sub-tropical environments (Cheung 1993; Narasimhan 1980; Rao et al. 1975; Rajagopal et al. 1998; Lee 1986). Given published growth rates were presented as a mix of growth per year and growth per month conversions were made to ensure uniformity across studies. Conversions to growth per month were achieved by dividing annual growth by 12 months, while growth per day values were based on annual growth divided by 365 days. Conversely conversion of growth month to growth year was multiplied by 12 months.

All published growth rates of *Perna viridis* in the scientific literature come from populations growing in tropical or sub-tropical (warm water) environments. The highest growth rate as reported by Rajagopal et al (1998) was 119 mm y^{-1}; however, this occurred under artificial conditions of high water velocity (i.e. on the heat exchangers of a coastal power plant. Under more natural conditions, maximal growth rates of 93 – 96 mm y^{-1} have been recorded (Narasimhan 1980; Rao et al 1975). These high growth rates tend to be associated with relatively high year-round water temperatures (26–32°C). Conversely, studies of *P. viridis* growth rates in Hong Kong, which has a lower minimum yearly water temperature (c. 12–17°C), recorded the lowest annual growth rates of between 49.7 and 60 mm y^{-1} (Lee 1986; Cheung 1993). It should be noted, however, that the growth rate studies of *P. viridis* in Hong Kong took place in polluted locations, which may have adversely affected overall growth rates. Thus using a range of growth rates should provide a basis from which to estimate age.

As the aim was to determine if the smaller sized cohort detected in the sea chest were a result of a spawning event while in WA waters age, estimates were focussed on this group. It is acknowledged that while these estimates do not provide an exact age they do provide a spawning window upon which decisions can be made. If estimates provide a window that states the sea chest cohort were likely to be from a spawning event elsewhere then this questions inspection protocols, if however, the window suggests that the sea chest cohort are likely to have spawned in WA then this stresses the need for more detailed and regular delimiting surveys.

The fastest and slowest published growth rates were used to estimate the age of *Perna viridis* collected from *HMAS Anzac* and *Arunta*. Estimates were based on the following assumptions: (1) that growth was linear and constant (as shown by Chatterji et al. 1984); and (2) that tropical growth rates can be used in the absence of temperate rates to provide a valid estimate of age. It is also acknowledged that the sea chest represents an artificial and modified habitat but, given the need to identify or clarify a potential spawning date for management purposes, we have worked within these assumptions. The smallest individual recorded in the sea chest of *HMAS Arunta* was 2.8 mm long and was estimated to be 8-20 days old (mean=11 days; Figure 3), while the largest animal from the sea chest was 52-126 days old (mean=71.6 days; Figure 3). As the vessels had been berthed for approximately six months at Garden Island, using the 'age window' provided by this range of growth estimates it is believed that the cohort detected within the sea chest on the port side of the vessel were the results of a spawning event that occurred while the vessel was berthed at *HMAS Stirling*.

Spawning and a recent 'heat pulse'

Using the documented growth rates and detection date a likely 'spawning' date of the juveniles from the sea chest was made. This spawning date ranged from 7 to 21 days for an animal of 3 mm and to between 35 and 107 days for an animal 15 mm in length. The mean length of juveniles collected from the sea chest of the *HMAS Arunta* was 9.4 mm long; using this size class a spawning date of between 21 and 64 days was calculated. This gives calendar dates of between 3rd February and 19th March 2011 (mean non heat pulse water temperature range 22.3 to 23.9 °C; heat pulse water temperatures 24.1-27.1°C).

Discussion

There is an increasing body of evidence that suggests increasing temperature is correlated with the growth and success of invasive species. The marine heat wave that occurred along Western Australia's coast created a body of water that raised ambient temperatures between 3 and 5 °C. In Cockburn Sound where the *HMAS Arunta* was berthed water temperatures for this February to March predicted 'spawning period' were up to 3 ºC higher than average (Marsh, unpublished data). While there is no direct proof of causality between the oceanic heat wave and the spawning of *Perna viridis* documented in this study there is a strong correlative relationship. The thermal shock experienced during the heat pulse is believed to be the trigger that initiated the spawning of this population. The spawning of *Perna viridis* in more tropical locals such as India has been linked to changes in water temperature (Rajagopal et al. 1988; Narisimham 1980). Shafee (1989) also report that in subtropical environs spawning in this species is restricted to warmer months. Hicks et al. (2001) report that increased water temperatures triggered spawning in *Perna perna*. Another species of *Perna*, *Perna canaliculus* is also reported to have increased rates of spawning associated with increased water temperature (Buchanan 1998). The increased reporting of invasive species settlement, growth and dominance linked to increasing temperature regimes add further weight to this hypothesis. Temperature is often thought of as one of the key limiting factors determining the geographic spread of many marine species. For tropical species such as *Perna viridis* it is predominantly colder waters that are perceived as a limiting factor. Perhaps one of the primary consequences of climate change will be the creation of suitable physical habitats for a non-indigenous species, such as *P. viridis,* to move outside its current native range through the removal of physiological (temperature) constraints in areas where it currently can not survive. A period of elevated sea temperature in the 1990s allowed a previously temperature-confined *P. viridis* population in Tokyo Bay to spread through Japan's Seto Inland Sea (Matsuyama 1999; Umemori and Horikoshi 1991). The average temperature ranges experienced within Cockburn Sound, Western Australia (14-24°C) are within the large range of documented thermal tolerances for *P. viridis* elsewhere (11-32°C NIMPIS, 2011). However, given that the adult population is proposed to have originated from tropical SE Asia (based on vessel movement history; with a sea temperature range of 28 - 31°C), it is unclear how well these tropical individuals would manage during the cooler times of the year (14-16°C). An understanding of the thermal tolerances, growth rates and reproductive potential of tropical pest species such as *P. viridis* in temperate waters may be important in light of the forecast changes associated with global climate change.

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