Understanding the Environmental Risks of Unplanned Discharges – the Australian Context: Benthic Macroinvertebrates

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Abstract

Marine invertebrates are sensitive to the toxic effects of oil. Depending on the intensity, duration and circumstances of the exposure, invertebrates can suffer high levels of initial mortality together with prolonged sublethal effects that can act at both the individual and population levels. Under some circumstances, recovery from these impacts can take years to decades. However the key impact mechanism is exposure and many factors can mitigate the degree of exposure (meaning that in many cases, impacts are moderate and recovery occurs within a few years). While a range of generalities can be stated about the response of marine invertebrates to oil spills, a key point is that almost every oil spill is unique in terms of its impact because of differing levels of exposure, as this is affected by a myriad of factors including: type and amount of oil, extent of weathering, persistence of exposure, application of dispersants or other cleanup measures, habitat type (including depth), species present and their stage of development or maturity, and processes of recolonisation, particularly recruitment. The importance of each of the factors and how they affect the degree of impact are explored in this review.

In the Australian context, the level of exposure to oil and subsequent impact will differ greatly between offshore wellhead accidents (e.g. Montara 2009), shipping accidents close to shore (e.g. *Pacific Adventurer* 2009), and refinery/oil storage depot spills (e.g. Port Stanvac 1999). Thus pre-spill planning and baseline assessment needs to be considered differently for each. Even though the volumes of oil are likely to be much greater from wellhead blowouts, the risk of direct impacts, at least to intertidal and shallow subtidal reefs and sedimentary habitats, may be lower than other types of spill. For this reason, pre-spill precautionary assessments should not just seek to establish baselines against which to assess impact, but should assess the risk to exposure of a range of oil-water fractions and hydrocarbon concentrations of marine invertebrates, and test the response of a range of marine invertebrate receptors to those concentrations. Both lethal and sublethal responses need to be assessed, and perhaps most importantly given the differing toxicities of different types of crude and refined oils, assessments need to be done using the oil most likely to represent greatest risk in terms of local geography.

Of the 44 significant oil spills in Australia since 1970, five occurred offshore with negligible likely or expected impact on benthic invertebrates. Despite the potential for oil spills to impact the marine environment, effects of only 21 of the spills were studied, and 18 had only cursory or no assessment of impact. Of the 21 spills with impact assessments, only 8 considered impacts on invertebrates, with many others focusing on the primary plant habitat affected but with little or no consideration of the invertebrate communities they support. With the exception of the 2009 *Montara* and 1999 *Torungen* spills, detailed assessment of spill impacts on invertebrates in Australia have been limited to temperate waters.

Very few assessments have considered the toxicity and sublethal effects of oil on Australian marine invertebrates. Those undertaken have been restricted to south-eastern Australia. While useful in the local context, responses of Australian marine invertebrates are needed across the range of habitats and geography and types of oil they might be exposed to. In particular, we identify a number of sessile habitat-forming filter-feeding invertebrates (sponges, bryozoa, tunicates) that urgently need assessment of their response to oil. In addition, more information is needed about Australian species for taxonomic groups that are known from overseas studies to be vulnerable to exposure to oil. These include molluscs, crustaceans and echinoderms. Lastly, in this review we outline some of the lessons learned in assessment of oil spill impacts from the studies examined, and provide some recommendations to be considered when responding to oil spills in Australia that are likely to affect marine invertebrates.

1. Introduction and Background

1.1 Background

Marine invertebrates comprise many groups of different organisms and occupy all areas of the water column from the sea surface to the seafloor and into the substrate. They include a highly diverse taxonomic range and include, for example, corals, worms, bluebottles, sponges, shells, sea urchins, starfish, crustaceans and nudibranchs. Their size ranges from microscopic organisms to several metres in length, and can vary incredibly in form.

Even though all marine ecosystems depend on invertebrates for their continued functioning, large gaps remain in our knowledge relating to their taxonomy, biology, ecological requirements and sensitivity to impacts; this includes marine invertebrate responses to the toxic effects of oil. Depending on the intensity, duration and circumstances of the exposure, invertebrates can suffer high levels of initial mortality together with prolonged sublethal effects that can act at both the individual and population levels. Despite the potential for impacts, however, this topic remains poorly studied and the majority of oil and fuel spills in Australian waters have had little assessment of possible or realised impacts of oil. Gartner et al. (2016) shows that of 44 spills since 1970, the potential direct effects of oil were studied in 21 cases, but only 8 cases considered impacts on marine invertebrates. Many studies focussed only on the primary plant habitat affected, e.g. mangroves, without considering the marine invertebrate communities they support (Allaway & Jackson 1979, Allaway 1982, 1985, Allaway et al. 1985, Allaway 1987, Burns et al. 1999). With the exception of the 2009 Montara wellhead spill and the 1999 Torungen spill, detailed assessment of spill impacts on invertebrates have been limited to spills in temperate waters (see Gartner et al. (2016) (this volume) for greater detail).

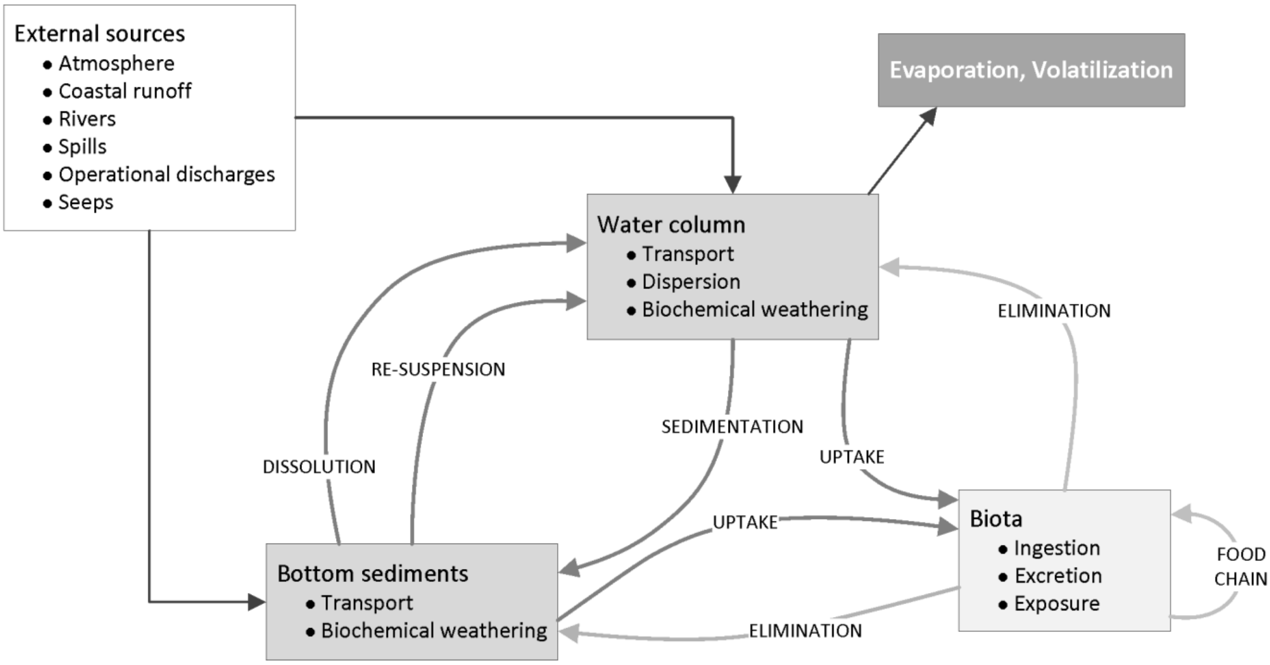
While this review attempts to consolidate what is known about the response of Australian benthic macroinvertebrate species to Australian crude oils, in general, this topic has been poorly documented (as evidenced by the extremely low availability of literature on the subject, described above). Furthermore many assumptions about Australian taxa must be based on extrapolation from studies carried out in other regions. Despite these limitations, however, this white paper reviews what has been learnt in the past two decades from laboratory studies, actual incidents and published literature on the effects of petroleum oils on marine invertebrates, and represents an update of what is known on the topic as relevant to the Australian context.

1.2 Behaviour and fate of crude oil in coastal environments

This review is principally concerned with the interaction between crude oils and marine benthic invertebrates within intertidal marine coastal areas. As such, it is important to understand how the behaviour and fate of crude oils vary in space and time, which in turn, can influence the likelihood and type of interaction spilt oils may have on benthic invertebrates.

Following a spill, it is expected that evaporation will generally remove about one-third of the volume of a medium crude oil slick within the first 24 hours, but there will always be a significant residue (National Research Council (NRC) 2003). Many crude oils will emulsify readily, a process that can greatly reduce subsequent weathering rates (NRC 2003). Crude oils also have the potential to adsorb onto intertidal sediments, with the risk of subsequent erosion of oiled sediments from the shoreline and deposition in near-shore habitats (NRC 2003). Dissolution from slicks and adsorbed oil can persist for weeks to years (NRC 2003). This is particularly relevant to understanding impacts on benthic invertebrates which principally occupy the seafloor and intertidal zones.

While near-shore oil spills obviously pose a high risk to benthic invertebrates, an offshore spill is much more likely to lose a significant portion of its toxic components before it reaches coastal waters or the seafloor (Haapkylä 2007), especially when dispersed. Most crude oils spread very thinly on open waters to average thicknesses of ~0.1 mm (Lee et al. 2013). The application of dispersants further enhances the transport of oil as small droplets into the water column. These can quickly entrain oil into the top 1 m of the water column and further dilute to concentrations less than 100 ppm when exposed to the turbulence of 1 m waves (Lee et al. 2013). Within 24 hours of discharge, it is expected that the dispersed oil will mix into the top 10 m of the water column and be diluted to concentrations well below 10 ppm, with dilution continuing as time proceeds. As biodegradation takes place over the following weeks, dispersed oil concentrations could be expected to decline to less than 1 ppm (Lee et al 2013).



Source: Amended from National Research Council (2003)

Figure 1.1 Conceptual model for the fate of petroleum in the marine environment

1.3 Environmental Protection and Biodiversity Conservation Act

The *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) is the key piece of Commonwealth legislation with relevance for the conservation of marine invertebrates. Under the EPBC Act, the protection of marine invertebrates is applied through a system of protected areas. These include the ability to declare marine protected areas in Commonwealth areas, the provisions relating to protection and management of World Heritage areas and RAMSAR wetlands of international significance, and the requirement to prepare impact assessments for Commonwealth-managed fisheries. Any activities must be referred to the Federal Minister for the Environment when they are likely to have a significant impact on these areas (as a Matter of Environmental National Significance (MNES)), include oil exploration and production.

1.4 This review

The aim of this paper is to review the impacts of oil and oil spills on marine invertebrates and to consider these impacts in an Australian context. Reviewing oil spill literature is a daunting task and there is always the risk of omission, for example, Harwell and Gentile (2006) reviewed almost 300 papers, just on the effects of the 1989 Exxon Valdez oil spill in Alaska. Other reviews on the effects of oil on marine invertebrates have been undertaken (Johnson 1977, Loya & Rinkevich 1980, Suchanek 1993), but no detailed review puts this topic into the Australian context.

Our assessment is limited to an evaluation of all of the small number of Australian studies that exist. We also draw on a subset of the international literature to capture the status of existing knowledge from field and laboratory studies and the experience of overseas oil spills, in order to identify the largest gaps in our ability to infer the likely impacts of future spills in Australia. In doing so, it became clear that much remains to be done to evaluate the risks of impact to marine invertebrates from unplanned oil spills. Regardless, this review has enabled us to draw conclusions and make recommendations about research that should be undertaken to achieve this. A need for further studies that test the response of Australian marine invertebrate species to impacts of oil and their risk of exposure is clearly apparent.

The scope for this review includes all life stages of marine invertebrates (except corals and other sessile cnidarians, which are dealt with in a separate white paper) with a sessile or motile benthic adult phase. We also include zooplankton, but we do not consider impacts on microbial communities. Other studies review and experimentally evaluate how microbial communities respond to oil (Macnaughton et al. 1999, Nyman & Green 2015). The complex nature of oil solubility and hydrocarbon fractionation in seawater, and thus the bioavailability of toxicants, make it difficult to compare results between studies using different oils and methods of creating water accommodated fractions (WAFs) (Redman 2015, Redman & Parkerton 2015), consequently we have not attempted to make such comparisons in reviewing literature in this paper. However, we emphasise that, in the Australian context, the circumstances and level of exposure are key elements needed when assessing risk, the likelihood of biological impact, and impact detection.

It is important to note that while Australia is expected to become the world's chief exporter of gas, very little is known about the environmental fate or impacts of gas condensate on marine invertebrates within the Australian environment (Hook et al. 2016), and therefore, this paper does not address this question. This paper also does not address impacts associated with oil spill clean-up operations, which while also relevant, are planned to be addressed in other APPEA publications.

Finally, it is also important to highlight that each uncontrolled oil release represents a unique set of physical, chemical, and biological conditions. As such, the information presented is a guide and not an absolute outcome, especially given the paucity of regional information in the Australian context.

2. Oil spills in Australia and the impact of oil on marine invertebrates - the Australian context

2.1 Offshore spills

Of the 44 spills listed in Gartner et al. (2016) (this volume), seven occurred far offshore from the mainland with only two of these having the potential to cause environmental impacts to benthic or intertidal habitats (the 2002 *Pacific Quest* event and the 2009 *Montara* platform wellhead blowout). Of these, only the *Montara* spill was subject to any detailed environmental assessment.

Assessments of the impact of the *Montara* wellhead blowout focussed on potential impacts to seabirds, reptiles, commercial fish species and coral reefs (PTTEP 2013). The nearest coral reefs (Vulcan shoals) were 27 km from the spill site, and assessments found no apparent recent mortality or disturbance impacts. However the assessments may have begun too late to detect impacts on invertebrate fauna associated with coral reef habitats. Most studies found the most significant mortalities were immediate, if they occurred at all. The *Montara* wellhead spill flowed for three months (21 August – 3 November 2009). Surveys of coral reef habitat began some eight months after the start of the spill, in April 2010 (Heyward 2010). The absence of any prior surveys or baseline data, and the delayed start of the post-spill survey, precluded the authors drawing firm conclusions about whether the *Montara* spill resulted in any impacts at Vulcan shoals or other surveyed locations (Heyward et al. 2010, Heyward 2011). The salient lessons from the *Montara* spill relevant to benthic macroinvertebrate assessment more generally include the need for baseline pre-spill monitoring in areas vulnerable to impact, and the necessity for a more rapid response to assess impacts. In addition, Heyward et al. (2010) found considerable ecological differences among shoals and reefs surveyed, indicating potentially different histories of disturbance. This would likely have confounded efforts to compare impacted and non-impacted reefs if there had been a significant effect of oil at one or more sites. This reinforces the need for any pre-spill baseline monitoring programs to include a historical series of measurements at reference sites not expected to be impacted in the event of a spill.

2.2 Sub-Antarctic spill

Australia has experienced one sub-Antarctic oil spill on Macquarie Island, from the *Nella Dan* in 1987, which resulted in significant impacts on marine invertebrates. These impacts have been well documented by Pople et al. (1990), Simpson et al. (1995), and Smith & Simpson (1995; 1998) who examined intertidal and subtidal habitats 1 year and 7 years post-impact (Pople et al. 1990, Simpson et al. 1995, Smith & Simpson 1995, 1998). Pope et al. (1990) also describe high mortalities of marine invertebrates immediately post-impact. These studies compared impacted and non-impacted (control) exposed rocky shores at different intertidal heights, as well as more sheltered kelp holdfast habitat. After one year significant differences existed between sites, with habitats at all impacted sites having lower abundances of marine invertebrates. Gastropods (limpets and trochids) and echinoderms (holothurians and sea stars) were heavily impacted on the exposed shores (Pople et al. 1990), while isopods were most impacted amongst more sheltered kelp holdfast habitats (Smith & Simpson 1995). After 7 years, invertebrate abundances at impacted sites on exposed shores were comparable to control sites, however significant differences remained within the more sheltered kelp holdfast habitat. Some impacted sites showed continuing presence of diesel and invertebrate communities dominated by opportunistic polychaete and oligochaete worms, whereas kelp holdfast invertebrate faunal assemblages at control sites were dominated by crustaceans such as isopods and other peracarids (Smith & Simpson 1998).

2.3 Spills in mangroves and estuaries

A moderate proportion (12/44) of spills in Australia have impacted mangrove habitats (Gartner et al. 2016) (this volume), however associated studies rarely considered impacts on fauna. After the small Arthur Phillip spill in 1985, Anink (1985) recorded up to 743 µg/g of hydrocarbons in sediment and significant impacts on mangroves in Botany Bay. They noted that invertebrate mortality (crabs) was not evident until ~4 weeks post-spill, and that the same observation had been made in the 1982 Parramatta River spill, with dead crabs first observed 3 weeks after the spill. Andersen et al. (2008) found evidence that high intertidal burrowing crabs in mangroves suffered high mortality (reduced incidence of crab holes) 1 month following the *Global Peace* spill in Gladstone harbour, but that after 6 months, crab hole numbers had recovered (Melville et al. 2009). Species richness and abundance of marine invertebrates in the lower intertidal zone did not differ between impacted and control sites 1 month post-spill.

2.4 Spills on temperate and sub-tropical shores and reefs

Following the 1995 Iron Baron spill in Tasmania, Edgar & Barrett (2000) undertook the first rigorous assessment of the impacts of an oil spill on a subtidal temperate reef in Australia. In contrast to the obvious mortality to high intertidal invertebrates (such as amphipods) caused from this spill, Edgar & Barrett (2000) found no evidence of oil impacts on species richness and abundance of reef invertebrates and fish across gradients of exposure to heavy oil mixed to sufficient depth to impact the reef. However, other Australian studies showed varying responses to oil spills when examining impacts at different intertidal levels. When assessing the impact of the *Pacific Adventurer* oil spill 1 week and 3 months post-spill, Schlacher et al. (2010) found significantly less diversity and abundance of invertebrates on impacted beaches low on the shore, with no detectable impacts on the upper shore. They found significantly less diversity and abundance on impacted beaches lower on the shore, with no detectable impacts on the upper shore. Three months post-spill, there was no evidence of oil remaining on the beach but the lower density and diversity of invertebrates on impacted beaches remained (Schlacher et al. 2010). For the same spill, assessment of impacts on rocky shores (Stevens et al. 2012) showed a very significant impact on diversity and abundance at both mid and high intertidal areas, with the most severe effects in the high intertidal. Significant effects remained five months post-spill; by four years post-spill, mid-shore communities of invertebrates had returned to pre-impact levels but higher intertidal communities continued to be affected (Finlayson et al. 2015).

MacFarlane & Burchett (2003) studied the impact of the 1999 *Laura D'Amato* oil spill on rocky intertidal reefs in Sydney Harbour. Pulmonate limpets (*Siphonaria*) and the trochid snail *Austrocochlea* suffered significant mortality at the most heavily oiled site. Less impacted sites also showed impacts of oiling on densities of some species, but these changes were of a similar magnitude to seasonal variability observed at some sites surveyed for 8 years prior to the spill. Their study in 2000 (MacFarlane & Burchett 2003) documented some recovery at all sites 12 months after the spill. A follow-up laboratory study (Reid & MacFarlane 2003) confirmed the toxicity of oil similar to that released from the *Laura D'Amato* to *Austrocochlea*. The same oil spill caused near total mortality of the amphipod *Exoediceros* on sandy beaches affected by the spill (Jones 2003). Rates of recovery tended to be fastest at the less impacted site, which recovered after 4 months compared to more heavily impacted sites, some of which showed no recovery after 9 months. Rapid recovery of communities was found by Dexter (1984), who documented a decline in polychaetes and amphipods on a beach impacted by a 1981 spill in Botany Bay, but with full recovery to pre-impact levels after 3–5 months.

2.5 Oil spill simulation field studies in Australia

Few studies simulating the effects of an oil spill on invertebrates in natural habitats have been undertaken in Australia. Clarke and Ward (1994) found applications of Bass Strait crude oil, crude oil plus dispersant, and diesel (which simulated an unplanned spill in salt marshes in Jervis Bay, NSW), caused high rates or mortality among gastropods (*Littorina*, *Bembicium*, *Salinator* and *Ophiocardelis*) and crabs. They found similar effects amongst the different contaminants used. Clarke & Ward (1994) found that treated plots had been recolonised from adjacent areas after 12 months. Residual lower densities in some treated plots after this time possibly resulted from greater predation of gastropods in the treated areas, where plant cover had been reduced by the simulated spill treatments.

McGuinness (1990) carried out similar experiments in both mangrove and salt marsh plots in Botany Bay, NSW using weathered (1 part oil agitated with 2 parts seawater) Dubai light crude oil. High mortality of the gastropods *Assiminea*, *Melosidula* and *Salinator* was measured, with increased mortality in salt marsh compared to mangroves. However plots recovered within weeks, due in large part to rapid recolonisation from adjacent habitats (McGuinness 1990).

Both these studies found that recovery was rapid due to recolonisation from adjacent plots. In the event of an actual oil spill affecting a large area of salt marsh or mangrove, recovery is unlikely to be as rapid or as effective as observed by McGuinness (1990) or Clarke & Ward (1994). In Gladstone, Queensland, Burns et al. (2000) treated plots of mangroves with Gippsland crude and Bunker C oils to compare the weathering effects of each, but did not examine the influence on mangrove invertebrates (Burns et al. 2000).

Thompson et al. (2007) conducted a study in Antarctica using synthetic lubricants to assess the effect of recolonisation in defaunated plots in marine sediments. He found that abundances of animals that had colonised the plots were the same 5 weeks post-oiling, but that community composition was quite different with numbers of certain crustaceans (amphipods, tanaidaceans and cumaceans) reduced in treated plots compared to controls (Thompson et al. 2007).

2.6 Measured responses of Australian fauna to oil toxicity in laboratory studies

Experimentally measured toxicities to oil and dispersed oil in Australia are limited to just a few studies. Gulec et al. (1997) measured the 96 hour 50% lethal concentrations (96 hour LC50s) for the amphipod *Allorchestes compressa* in WAFs of 311,000 ppm, 16.2 ppm and 14.8 ppm Bass Strait crude oil and crude oil plus dispersants (Corexit 9500 and Corexit 9527), respectively, indicating that the oil alone (without dispersant) was much less toxic. They also demostrated that a WAF using burnt oil was less toxic than unburnt oil (Gulec & Holdway 1999). *Octopus pallidus* hatchlings exposed to WAFs of Bass Strait crude oil and dispersed oil had a 48 hour LC50 of 0.39 ppm and 1.83 ppm, respectively, suggesting that they are much more sensitive to oil than amphipods (Long & Holdway 2002). The trochid snail *Austrocochlea porcata* had a 96 hour LC50 of 12 ppm (Reid & MacFarlane 2003). Although not conducted on Australian invertebrate species, Neff et al. (2000) compared the toxicities of three types of light to medium density north-west Australian crude oils and diesel fuel on penaeid prawns, mysid shrimps and sea urchin larvae (Neff et al. 2000) (see Section 3.3.2).

Unfortunately, studies of experimental laboratory (Section 2.5) and field-based oil toxicity are not directly comparable given the use of different field/laboratory conditions, weathering techniques (with/without pre-distilling, and WAF varying between 2 and 9 parts seawater), LC50 calculations (ppm total hydrocarbons vs % WAF), and time periods (24–96 hours). The use of regression analyses to predict LC50s from experimental data also makes comparison difficult, and applications to field situations even more challenging. As described above, Long & Holdway (2002) calculated a 48 hour LC50 for newly hatched octopus of 0.39 ppm total hydrocarbons. However they also reported a 48 hour no observed effect concentration of 0.36 ppm, and a lowest observed effect concentration of 0.71 ppm. Given that exposure times and circumstances in the field can vary greatly, the approach taken by Long & Holdway (2002) of also reporting 24 hour LC50s is useful when trying to extrapolate likely ecological field effects from laboratory toxicity experiments. These points highlight the need for ecologically meaningful approaches, and improved consideration of how laboratory toxicity trials are related to the field situation.

Three Australian studies have found sublethal behavioural impacts on seas stars and a gastropod. Ryder et al. (2004) found that the herbivorous sea star *Patiriella (now Parvulastra) exigua* from Port Phillip Bay avoided oiled sediment in the laboratory, and in doing so was able to avoid its narcotising effects (Ryder et al. 2004). The ability of the predatory Port Phillip Bay sea star *Coscinasterias muricata* to locate prey was significantly reduced when exposed to WAFs of Bass Strait crude oil with and without added dispersant; however sea stars exposed to a burnt oil WAF maintained the same ability to locate prey as control animals (Georgiades et al. 2003). Gulec et al. (1997) studied suppression of burying behaviour of the marine sand snail *Polinices conicus* after 30 minutes exposure to WAFs of Bass Strait crude oil alone, crude oil plus Corexit 9500, and crude oil plus Corexit 9527. Burying was suppressed in 50% of snails (EC50) at 190,000 ppm, 65.4 ppm and 56.3 ppm, respectively (Gulec et al. 1997).

3. Impacts of oil on marine invertebrates

3.1 Toxic impacts of oil

3.1.1 Impacts on zooplankton and larval stages

Larval life stages of marine invertebrates are likely to represent the most vulnerable period for exposure to toxicants. However, in long lived iteroparous species with short larval periods, the impact of a hydrocarbon spill on a population or species may be minimal. On the other hand, semelparous species with long larval periods might suffer a major impact.

Larval assays of toxicity to contaminants in seawater have been used extensively to determine both the toxicity of different oils and their fractions, and what concentrations and time exposures constitute lethal or sublethal or minimum observable effects. Sea urchin and bivalve mollusc larvae have been most commonly used, as they are easy to culture or are commercially available. Assays have used either contaminated seawater collected following a spill, or the preparation of known concentrations in the laboratory. The former method is likely to give results that can be directly related to a spill event, for example, Beiras and Saco-Alvarez (2006) used seawater sampled from the shore at various times following the *Prestige* oil spill in Spain in 2002 to test for toxicity against sea urchin, *Paracentrotus lividus*, larvae. They found that even after a four-fold dilution this WAF was toxic to the larvae immediately after the spill and that toxicity to the larvae from the undiluted contaminated seawater persisted even two months later.

Generally speaking, the planktonic larvae of marine invertebrates are highly sensitive to the toxic effects of hydrocarbons in a WAF. Chia (1971) reported on an oil spill in northern Washington State, USA, which killed numerous adult marine invertebrates and noted that the spill of diesel had occurred during the spawning season for many of them. He then tested the larvae of 14 species of echinoderms (sea stars and sea urchin), gastropod, bivalve and chitons molluscs, annelid worms, and a barnacle and an ascidian in a 0.5% oil water mixture and found that while all larvae in control (no oil) conditions survived, all larvae in the WAF died within 3 hours to 3 days except one sea star *Crossaster* which had all died after 8 days (Chia 1971). Unlike the other invertebrates, *Crossaster* has large yolky eggs with lecithotrophic (non-feeding) development, suggesting such animals might be more resilient to oiling. Around the same time, Wells (1972) also demonstrated the toxic affect of oil on lobster (Crustacea) larvae (Wells 1972, Wells & Sprague 1976, Stejskal 2000). Byrne and Calder (1977) and Nicol et al. (1977) further demonstrated that oil disrupted embryonic development causing mortality in a bivalve mollusc and a sea urchin, while PAHs have been demonstrated to inhibit settlement of sponges or incur mortality of sponge recruits (Cebrian & Uriz 2007). For zooplankton, Elmgren et al. (1983) found ostracods and harpacticoid copepods were significantly impacted by the 1977 *Tsesis* spill in Spain. Almeda et al. (2013) examined the effects of crude oil on mortality on a wide range of copepod species in the Gulf of Mexico and determined an LC50 of just 31.4 µL/L WAF. In other studies, Almeda et al (2014a; 2014b) demonstrated that dispersed oil is more highly toxic than crude oil alone. This issue is discussed further in Section 3.3.3.

Following the Gulf of Mexico Deep Water Horizon spill in 2010, settlement of the commercially important crab *Callinectes* along the Mississippi coast was measured (Fulford et al. 2014). Natural settlement rates of this species varies considerably each year, however they did not detect any change that could be attributed to the oil spill. They also noted that although the oil is toxic to crab larvae at certain concentrations, those concentrations were not experienced in the areas important for crab settlement in their study area.

It is evident from studies that the impact of oil on larvae is largely dependent on the exposure concentration (Almeda et al. 2013, Fulford et al. 2014). Thus, the risk of exposure to toxic concentrations is a critical element in the evaluation of environmental risk. This emphasises the importance of studies that determine at what concentrations hydrocarbons cause mortality and significant sublethal impacts to receptor species, and the likelihood of exposure to those concentrations. Because of the varying responses observed by different species to different types of oils, and the degree of weathering, the situational context becomes very important and underlines the importance of studies that not only provide data in a context relevant to Australia but do so in at least a regional or preferably local context. For this reason, we advocate tests that determine the toxicity of Australian oils against taxa most likely to be exposed to them. In addition we suggest that tests seek to simulate the concentration levels likely to occur at the time of exposure. This is especially emphasised for pre-spill assessments for wellheads, where an advancing plume would create an exposure risk to animals in intertidal and subtidal habitats.

3.1.2 Impacts on adults

Early reports on the impacts of oil spills were generally from intertidal exposures following shipping or refinery accidents, and indicated a very wide range of marine invertebrate taxa are affected by oil, and that mortalities can be very high (Mitchell et al. 1970, Spooner 1970, Chia 1971, Woodin et al. 1972, Chan 1977). Given the different taxa that characterise species assemblages across the wide range of benthic habitat types and latitudes, we have sought to give examples below of the information available on different types of marine invertebrates and how they are affected by exposure to oil.

Sponges

Sponges form an important and often dominant component of the fauna of many Australian benthic marine habitats (Fromont et al. 2012), and this is especially true of the Australian North West Shelf (NWS) where extensive oil and gas exploration and production is occurring. There are few reports of sponge mortality from oil spills in the international literature, but those that do exist suggest that they may be highly vulnerable to oil toxicity. After the 1986 oil spill in Galeta, Panama, sponges growing on oil-covered mangrove roots died (Burns et al. 1993). Similarly, following the 2002 *Prestige* oil spill *Hymeniacidon perlevis* and *Tethya* sp. (defining species in the lower intertidal area of their study site in France) were killed and had not reappeared despite annual monitoring up until 2011 (Castège et al. 2014). Batista et al. (2013) determined that *Hymeniacidon heliophila* was a good indicator of polycyclic aromatic hydrocarbon (PAH) pollution in Brazil (Batista et al. 2013). However, apart from the study of Harvey et al. (1999), who found an absence of genotoxic effects on *Halichondria panicea* after the 1996 *Sea Empress* spill in Wales (Harvey et al. 1999), we are not aware of any laboratory studies on the toxic effects or harmful concentrations of oil to adult stage sponges. This is arguably a high priority need in the Australian context.

Bryozoans

Bryozoans can comprise a significant proportion of benthic biomass, and this is particularly true on Australia's NWS (Keesing et al. 2011). However this group is rarely reported when considering the impacts of oil spills, although Burns et al. (1993) found that along with hydroids, bryozoans were the least impacted and the fastest taxa to recover on the roots of mangrove trees following the 1986 Galeta, Panama refinery oil spill. In the absence of any other studies, examining the response of Australian bryozoa to oil is important.

Cnidarians

As a group, anthozoan sceractinian corals are sensitive to oil pollution and can suffer high mortality on both intertidal and subtidal reefs affected by oil spills (Jackson et al. 1989), as well as chronic sublethal effects (see reviews by Johnson 1977; Loya & Rinkevich 1980; Suchanek 1993) – this group is covered in detail in Westera (2016).

The reported responses of other cnidarians to oil vary considerably in the literature, but as a group they are probably more resilient to the effects of oil than most other groups of marine invertebrates. Jackson et al. (1989) recorded that the hydrozoan *Millepora* and zoanthids *Palythoa* and *Zoanthus sociatus* were significantly affected (along with scleractinian corals) after the 1986 Galeta, Panama refinery spill. Cohen et al. (1977) found that toxicity of crude oil on octocorals (*Heteroxenia fuscescens*) was only evident at very high levels of exposure (12 ppt), but that sublethal effects occurred at lower concentrations (Cohen et al. 1977). There is some evidence that hydroids may be resilient to oil spill impacts, but this is equivocal. Suchanek (1993) reviewed laboratory studies that indicated the hydroid *Tubularia* and the scyphozoan *Aurelia* were both sensitive to oil; conversely, Burns et al. (1993) found that hydroids (and bryozoans) growing on mangrove roots were minimally impacted following the 1986 Galeta, Panama refinery oil spill.

Anthozoan actinians (anemones) were severely impacted by the 1986 Galeta oiling and much slower to recover than other taxa, with reduced densities even after 5 years (Burns et al. 1993). However Castege et al. (2014) found two anemones (*Actinia equina* and *Anemonia viridis*) were among a group of invertebrate species that were minimally impacted or recovered quickly (within one year) following the 2002 *Prestige* oil spill that affected the French coastline. Similarly, the anemone *Anthopleura elegantissima* was one of the few species that survived the 1957 *Tamico Maru* spill in Mexico that killed the majority of marine invertebrates (Mitchell et al. 1970). The widely varying responses of cnidarians, and their importance among benthic marine communities in Australia (and especially the tropics; e.g. Keesing et al. 2011), indicates that a specific examination of their response to expected levels of exposure in Australia is required.

Crustaceans

Motile crustaceans as a group are among the most vulnerable marine invertebrates to oil spills and suffer high mortalities, behavioural disorders, and reduced recruitment (Krebs & Burns 1977) Crabs are highly conspicuous components of intertidal assemblages, and are among the first casualties to be reported after a spill (e.g. Spooner 1970; Woodin et al. 1972; Chan 1977). Following the 1969 spill in West Falmouth, Massachusetts, USA, fiddler crabs (*Uca punax*) suffered high mortality. Amphipods, isopods and crabs were heavily impacted by the 1978 Amoco Cadiz spill in France (Chassé 1978, O'Sullivan 1978, Conan 1982) while stomatopods were also heavily impacted in intertidal seagrass beds after the 1986 refinery spill at Galeta, Panama (Jackson et al. 1989). Following the 1977 *Tsesis* oil spill in the Baltic Sea, numbers of amphipods (*Pontoporeia*) were reduced by 95% as oil began to be deposited onto the benthos (Elmgren et al. 1983).

Although high initial mortalities can occur following spills, some crustaceans apparently recover quickly. After the 2002 *Prestige* oil spill in France, several shrimp and crab species (*Athanas nitescens*, *Carcinus maenas*, *Eriphia spinifrons*, *Galathea squamifera*) were among a group of invertebrate species on a rocky intertidal shore that were minimally impacted or recovered quickly (within one year; Castege et al. 2014). Conversely, after the West Falmouth spill the fiddler crab population had still not recovered seven years post-spill (Krebs 1977).

Compared with motile crustaceans, adult barnacles as a group have been regarded as very resistant to the effects of oil (Suchanek 1993). Following the *Amoco Cadiz* spill, Chasse (1978) found barnacles *Chthamalus* and *Balanus* did not suffer mortality, although this may have been due to their position at a lower intertidal height on the shore, as extensive gastropod mortalities were found higher on the shore. In Brazil, Lopes et al. (1997) studied the impacts of an oil pipeline spill and found that among the crustaceans, crabs and isopods suffered heavy mortality but barnacles (*Chthamalus* and *Tetraclita*) were not significantly affected (Lopes et al. 1997). However, the circumstances of a spill (and not the type of animal) are probably more important in determining the levels of mortality experienced. For example, following the 1971 diesel spill in Washington state, USA, substantial mortality of *Balanus glandula* and *Balanus cartosus* were recorded (Woodin et al. 1972), while after the 2002 Prestige spill in Spain, mortality of the barnacle *Chthamalus montagui* depended on extent of oiling (Penela-Arenaz et al. 2009).

Some Australian studies have reported significant oil spill impacts on amphipods (Edgar & Barrett 2000, Jones 2003) and crabs (Anink 1985, Clarke & Ward 1994, Andersen et al. 2008) from accidental or planned oil spills in Sydney Harbour, Jervis Bay and Tasmania. Given their vulnerability to oil and the commercial importance of crustacean invertebrates, more attention should be given to assessing the toxicity and sublethal responses of crustaceans to oil in areas of anticipated risk across different Australian regions.

Tunicates

Tunicates were among the heavily impacted taxa within the invertebrate communities on the roots of mangroves subject to oiling following the 1986 refinery spill in Galeta, Panama. Like anemones, tunicate populations had not recovered after 5 years (Burns et al. 1993). Castege et al. (2014) reported a similar time (2–5 years) for the tunicate *Botryllus schlosseri* to reappear at their study site in France after the 2002 *Prestige* oil spill. There are few reports of the impacts of oil on tunicates. Nevertheless, given their importance as filter feeders and the abundance of some species (such as *Pyura stolonifera*) among intertidal and benthic assemblages in eastern Australia (Dakin 1960), and among subtidal habitats on the NWS (Keesing et al. 2011), more work on their vulnerability to oil spills is warranted.

Worms

We group a diverse range of worm-like phyla here, and not surprisingly they have a diverse range of sensitivities to oil. In his review, Johnson (1977) considered several studies and concluded that adult polychaetes were in general highly resistant to oil toxicity. The polychaete *Capitella capitata* opportunistically proliferates in anthropogenically disturbed sediments, including those impacted by oil, even where very high mortality of other invertebrates occurs. An extreme example of this is described by Sanders (1978) after the Florida oil spill in West Falmouth, USA (Sanders 1978).

However, some polychaetes have been reported to suffer significant mortalities following oil spills, including after the *Amoco Cadiz* spill in France (O'Sullivan 1978) and the 1974 Bouchard spill in Massachusetts, USA, where a large number of *Nereis virens* were killed among numerous other marine invertebrates (Hampson & Moul 1978). Elmgren et al. (1983) reported polychaetes (*Harmothoe sarsi*) were reduced by 95% by sedimented oil from the 1977 *Tsesis* oil spill in the Baltic Sea, and that turbellarians and kinorhynchs were also significantly affected, but that nematodes were not greatly affected. Beyrem et al. (2010) examined the response of lagoon sediment nematode assemblages from Tunisia to lubricating oil contamination in laboratory experiments, and found this caused reductions in both abundance and species diversity although individual species responded differently. Some like *Daptonema trabeculosum* were very sensitive to the oil while *Spirinia gerlachi* was resilient; the reason why this disparity occurred between taxa was not established (Beyrem et al. 2010). Nematodes in deep water (~1200 m) samples were found to respond positively to the 2010 *Deep Water Horizon* spill in the Gulf of Mexico (Montagna et al. 2013). It was hypothesised that the nematodes may have responded positively to enhancement of the bacterial flora through oil-induced organic sediment enrichment and reduction of competitive species in taxa such as copepods that were negatively impacted by the spill. Dexter (1984) reported reductions in polychaetes after an oil spill in Australia, but in general knowledge of the likely impacts of oil spills on worms in the Australian situation is not known.

Echinoderms

Echinoderms are among the most vulnerable of marine invertebrates to oil spills, and many early studies that documented oil spills indicated extensive mortalities of echinoderms after a spill (e.g. Mitchell et al. 1970; Chia 1971; Woodin et al, 1972; Chan 1977; Jackson et al. 1989). Castege et al. (2014) found three species of echinoderms (the ophiuroid *Amphipholis squamata* and the echinoids *Echinus esculentus* and *Psammechinus miliaris*) were among a group of invertebrates that disappeared from a French rocky shore after the 2002 *Prestige* oil spill, and took 2–4 years to recover. On the other hand, the sea star *Asterina gibbosa* and a holothurian (*Holothuria* sp.) were minimally or not impacted. Conan (1982) reported severe mortality of one million heart urchins following the 1978 Amoco Cadiz spill in France. Ballou et al. (1989) simulated the effects of crude oil and dispersed oil on a coral reef in Panama, and found that all *Echinometra lucunter* and *Lytechnius variegatus* were killed in the experimental treatment areas (Ballou et al. 1989). Jackson et al. (1989) also reported high mortality (~80%) of *Echinometra lucunter* in Panama. Echinoderms also experience a range of significant sublethal impacts from oil exposure on their movement, reproduction and feeding (see Section 3.2 and review by Johnson 1977).

Australia's marine ecosystems harbour a high level of echinoderm diversity, including in areas where petroleum exploration and extraction, as well as shipping, are important. For example, there is a species of sea star in the Great Australian Bight (*Parvulastra parvivipara*) that has a species distribution of < 200 km (Edgar 2012), which is very small for a marine species. On the NWS, ecologically important heart urchins can be superabundant (Keesing & Irvine 2012). The density and diversity of crinoids on the Great Barrier Reef is extraordinarily high (Bradbury et al. 1987), and yet we could find no studies on oil toxicity to crinoids anywhere. There have been two studies on sublethal impacts of oil on sea stars in south-eastern Australia (Georgiades et al. 2003, Ryder et al. 2004), but no tropical studies or studies on other classes of echinoderms – this is an important priority for future work.

Molluscs

Gastropods, particularly herbivores, are consistently reported as experiencing very high mortality from oil spills (e.g. (Mitchell et al. 1970, Woodin et al. 1972, Chia 1973, Le Hir & Hily 2002), with mortality rates dependent in large part on degree of exposure, which in turn is often dependent on shore height in intertidal populations. Following the 1978 *Amoco Cadiz* spill in France, Chasse (1978) and O'Sullivan (1978) documented high mortality of gastropods *Littorina*, *Gibbula* and *Monodonta*, and to a lesser extent *Patella* limpets. In that study, mussels escaped mortality due to their position lower in the intertidal zone where barnacles were also unaffected. However in other studies where there has been heavy oiling of mussels, they have also been shown to suffer high mortality (e.g. Mitchell et al. 1970). In the *Amoco Cadiz* spill, Conan (1982) refers to the massive mortality of 14.5 million bivalves of other families (Cardiidae, Solenidae, Macridae and Veneridae).

Following the 1971 diesel spill in Washington state, USA, Woodin et al. (1972) recorded substantial mortality of numerous molluscs including chitons (*Mopalia* sp. and *Katharina tunicata)*, bivalves (*Clinocardium nuttalli* and *Macoma* spp.) and gastropods, (*Acmaea* spp.), while oysters (*Crassostrea gigas*), mussels (*Mytilus edulis*) and the gastropods (*Littorina scutulata* and *Littorina sitkana*) experienced little or no mortality. Predatory whelks (*Thais* spp.) were found that appeared moribund, but recovered when returned into clean seawater. In another study on the same spill, Chia (1973) found extensive mortality of marine invertebrates including limpets and chitons, but noted that two species of periwinkles (littorinids) seemed unaffected. Conversely, following a large 1986 refinery spill in Panama, Garrity & Levings (1990) found that both neritids and littorinids were severely impacted and almost absent from the affected sites for more than two years, although the severity of effects on molluscs (in terms of immediate mortality) varied spatially with the amount of oil deposited (Garrity & Levings 1990). Subtidal impacts on gastropods (e.g. mortality of the abalone *Haliotis rufescens* and other subtidal gastropods) have also been reported after the *Tampico Maru* tanker was shipwrecked and spilled oil for 8–9 months in Baja California, Mexico in 1957 (North et al. 1965, Mitchell et al. 1970).

Some molluscs have been reported to be resilient to the effects of oil. For example, *Cerithium* gastropods have been reported to continue to feed on oiled intertidal flats (Spooner 1970, Chan 1977). As a result of the 1977 *Tsesis* oil spill in the Baltic Sea, Elmgren et al. (1983) showed that despite sedimented oil killing 95% of amphipods and polychaetes, the clam *Macoma balthica* experienced minimal mortality despite becoming highly contaminated (to a level of 2 mg/g of total hydrocarbons).

As toxic PAHs can readily bind to sediment, phytoplankton and other particulate organic matter, they can be readily ingested by filter feeding invertebrates such as sponges, mussels and oysters. Bivalve molluscs in particular are effective at bioaccumulation of these toxicants (La Peyre et al. 2014), and so can potentially be subject to a range of sublethal impacts. Given the range of often contradictory and inconsistent responses of molluscs to oil, and their diversity and importance in intertidal and subtidal assemblages around Australia, there is a need for studies on the response of local species to the types of oil they may be exposed to in different parts of Australia.

3.2 Sub-lethal, chronic and indirect effects

3.2.1 Impacts on reproduction

Reduction in the success or extent of reproductive activity following exposure to oil has been shown for several different invertebrate types, and indicates that sublethal effects of oil can threaten reproductive success of a population impacted by oil.

Berdugo et al. (1977) showed a reduction in fecundity, brood size and rate of egg production in planktonic copepods following exposure to oil (Berdugo et al. 1977). Similarly, sublethal concentrations of crude oil were found to decrease brood numbers in amphipods when females were exposed during the incubation period (Linden 1976). Elgrem et al. (1993) also found that several months following the *Tsesis* oil spill in Spain, female amphipods (*Pontoporeia affinis*) showed a significant increase in abnormal eggs.

Blumer et al. (1970) found that while an oil spill did not cause mortality to mussels, they failed to reproduce following the spill (with mussels from an area not affected by the oil spill reproducing normally) (Blumer et al. 1970). Other studies (Renzoni 1973, 1975, Nicol et al. 1977) showed that exposure to No. 2 fuel oil WAF affected sperm motility and reduced fertilization in sand dollars and bivalves. Vashchenko (1980) reared sea urchin larvae from gametes obtained from adult *Strongylocentrotus nudus* maintained for 45 days in sea water containing 30 mg/L of diesel. While larvae from control urchins developed normally, those produced from gametes of urchins maintained in the diesel-contaminated water resulted in a high proportion of abnormal and non-viable larvae after 3 days, including those larvae reared from control eggs or sperm and treatment eggs or sperm (i.e. if one set of gametes came from a control urchin, the larvae still did not develop normally) (Vashchenko 1980).

Karinen et al. (1985) exposed Dungeness crabs (*Cancer magister*) to various concentrations of crude oil contaminated sediment and found that moulting was affected, mating was often unsuccessful, and that egg-carrying females produced significantly lower numbers of larvae than control crabs. These larvae also had shorter survival times than larvae from control crabs (Karinen et al. 1985).

3.2.2 Impacts on movement, attachment and feeding

Animals not exposed to oil at concentrations high enough to kill them may still suffer mortality directly related to the oil impact. For example, a reduced ability to move away from oiled areas or escape predators may reduce survival. Percy and Mullin (1977) found impaired movement in amphipods *Onisimus affinis* and hydromedusa *Halitholus cirratus* when exposed to low concentrations of crude oil. Johnson (1977) provides numerous examples of the narcotising effect of hydrocarbons in oil causing reduced mobility and respiration in decapod crustaceans, leading to reduced survival and increased vulnerability to predation (including during the process of moulting). O'Sullivan (1978) found that after the 1978 Amoco Cadiz oil spill in France, limpets remain attached to the reef but their grip was weakened. This may have affected their ability to survive strong breaking waves or resist predators. Similarly, Mageau et al. (1987) demonstrated impairment of movement and attachment in the urchin *Strongylocentrotus droebachiensis* from loss of movement of tube feet and spines following exposure to dispersed crude oil in an experiment in the Arctic.

Numerous studies have shown that sublethal exposures of marine invertebrates to oil result in reduced feeding rates or ability to feed effectively. Feeding rates of the predatory sea star *Asterias rubens* on mussels were depressed when exposed to crude oil at 200 ppm (Crapp 1971). A similar reduction in feeding rate and a reduced growth rate was found in *Evasterias troschelii* feeding on mussels in Alaska after the sea stars had been exposed to a very dilute (0.12 ppm) crude oil seawater mixture (O'Clair & Rice 1985).

3.2.3 Diseases

A range of tumour and blood type diseases of crustaceans and molluscs have been reported in animals exposed to oil spills (see (Hodgins 1977)) for review of oil-induced disease responses in fish and invertebrates). DNA damage and potential mutations from oil exposure were found in mussels following the *Prestige* oil spill in Spain (Perez-Cadahia et al. 2004, Laffon et al. 2006) and the Aegean Sea oil spill in Spain (Sole et al. 1996). However after the 1996 *Sea Empress* spill in Wales, DNA damage and potential mutations were found to be more prevalent in fish than invertebrates (mussels and sponges) (Harvey et al. 1999).

3.2.4 Changes to community assemblages and trophic structure

The sudden loss of a particular group of animals in any disturbed habitat often leads to a change in community structure caused by founder effects, and this has certainly occurred after some oil spills. Numerous studies following oil spills report a positive response by, and occasionally a proliferation of, marine algae (Mitchell et al. 1970, Chan 1977, Chassé 1978, Southward & Southward 1978, Newey & Seed 1995, Le Hir & Hily 2002, Marshall & Edgar 2003, Barillé-Boyer et al. 2004) an outcome attributed either to a loss of herbivores or to an increase in nutrients (or both). Following the 1993 *Braer* oil spill in Scotland, Newey and Seed (1995) found a proliferation of *Ulva* and *Porphyra* in the impacted area where grazers (such as limpets) were greatly reduced. Castège et al. (2014) found an overshoot recovery of grazers ~18 months after the 2002 *Prestige* oil spill in France, and this was attributed to rapid algal growth in the oil-impacted area. Also within France, a dramatic increase in algae (*Ulva* and *Grateloupia*) in intertidal rock pools after the 1999 *Erika* oil spill was recorded, which resulted in 100% mortality of the grazing sea urchins *Paracentrotus lividus* and *Psammechinus miliaris* (Barillé-Boyer et al. 2004). In some of these cited cases, the algae was able to respond not just to a reduction in grazers, but also to an abundance of bare spaces caused by the death of encrusting cover invertebrate species such as barnacles (Newey & Seed 1995) and sponges (Castège et al. 2014).

The selective nature of oil toxicity caused by the resilience of some groups can also alter community structure. Following a large 2006 oil spill off Estonia, Kotta et al. (2008) compared the abundance of guilds of herbivores, suspension feeder and deposit feeding invertebrates immediately after the spill and 18 months later, and found that herbivores (especially amphipods and isopods) were decimated while deposit feeders and suspension feeders were not impacted. Three years after the 1978 *Amoco Cadiz* spill in France, Conan (1982) found that opportunistic polychaetes had come to dominate sand/mud habitats but that clam populations had not recovered and had unstable recruitment. As well as selective mortality affecting communities after an oil spill, opportunistic recruitment by less sensitive species can affect community dynamics. For example after the 1989 Exxon Valdez spill in Alaska, Jewett et al. (1999) found the abundance and biomass of subtidal epifauna and infauna at oiled sites among seagrass beds was higher than at control sites, partly due to the response of mussels and polychaetes to organic enrichment at the oiled sites. These effects were found to persist for at least 6 years.

Peterson (2001) and Peterson et al. (2003) documented the long terms effects of the 1989 *Exxon Valdez* spill and concluded that profound chronic effects remained more than 10 years later, particularly in trophic interactions and effects on populations of birds and sea otters, some of which related back to their invertebrate prey or foraging areas (Peterson 2001, Peterson et al. 2003). Suspension-feeding clams and mussels can only slowly metabolise hydrocarbons, and when continuously exposed to sedimented oil they concentrate the hydrocarbons, leading to chronically elevated tissue contamination (Peterson et al. 2003). In the case of the 1989 Exxon Valdez spill, persistent bioaccumulation of hydrocarbons within clams (*Protothaca staminea*) and mussels (*Mytilus trossulus*) meant that foraging sea otters that consumed the bivalves suffered chronic exposure to hydrocarbons for many years (Peterson et al. 2003). Carls et al. (2001) estimated it could take 30 years for mussel beds to be free of hydrocarbon contamination because of oil trapped in the sediments beneath the beds (Carls et al. 2001). However Payne et al. (2008) argued (on the basis of samples collected 11 years longer than Carls et al. 2001) that levels of contamination in mussels had reduced to a very low levels by 2006, and that bioaccumulation by mussels only presented a problem when substrate with sedimented oil was disturbed (Payne et al. 2008). A continuing problem is that foraging birds and sea otters often disturb sediment in the Exxon Valdez spill area, exposing the sedimented oil (Peterson et al. 2003, Payne et al. 2008).

3.2.5 Loss of genetic diversity

Loss of genetic diversity in the razor clam *Ensis siliqua* after four years was attributed to the 2002 *Prestige* oil spill in Spain (Fernandez-Tajes et al. 2012). These authors compared pre-spill data of genetic diversity with that from the population that recovered following the spill. The latter having been produced from a very much smaller population than existed before the spill. This result contrasted that of Pineira et al. (2008), who after the same spill did not find any evidence for reduced genetic diversity in the littorinid snail *Littorina saxatilis*. The difference in results are probably explained by the short time (only 18 months) after the spill that Pineira et al. (2008) made their study and/or the widely differing life histories of the two molluscs. *Littorina saxatilis* is ovoviviparous, brooding its young internally before hatching, and the species has very limited dispersive capacity, while *Ensis siliqua* is a free spawning species with external fertilisation. The latter life history strategy should permit greater gene flow within and between populations but this did not occur in this case, at least in the initial years following the spill.

3.3 Factors influencing impacts

3.3.1 Impacts of residual oil

One important mechanism for prolonging the effects of an oil spill is sedimentation. Oil is readily sedimented into muddy or sandy substrates, where it can both restrict the recovery of infauna and burrowing fauna (such as crabs) and cause secondary continuous exposure via erosion of the sedimented layers (e.g. (Hayes et al. 1993)). Burns et al. (1993) found that five years after the 1986 refinery oil spill in Galeta, Panama, sedimented oil was still leaching from mangrove sediments and continued to bioaccumulate in bivalve molluscs.

Sedimentation of oil also has an effect on re-colonisation and bioturbation. Wells & Sprague (1976) found that sedimented oil disrupted the burrowing behaviour of post-larval American lobster *Homarus americanus*. Dow (1978) showed how five successive year classes of the burrowing bivalve *Mya arenaria* were killed following the 1971 Long Cove oil spill in Maine, USA, as new recruits burrowed into sediment contaminated to 250 ppm oil at 15–25 cm below the surface. Gilfillan and Vandermeulen (1978) found the same species was still subject to significant lethal and sublethal effects six years after the 1970 *Arrow* oil spill contaminated lagoon sediments (87–3800 ppm) in Nova Scotia, Canada, with reduced growth and metabolic rates and fewer mature adults than at an unimpacted site. In Argentina, Ferrando et al. (2015) used cores extracted from muddy Argentinean sediments to show how oil contamination results in reduced bioturbation following the mortality of infaunal species. This effect will exacerbate oil spill impacts, by reducing irrigation and oxygenation of subsurface layers, resulting in anoxic effects as an indirect effect of oiling.

3.3.2 Composition of oil and weathering

It is apparent that refined oils, diesel and heavy bunker fuel oils are more toxic than crude oil. Anderson et al. (1974) compared the toxicity of a heavy fuel oil (Bunker C), a light fuel oil (Number 2, similar to diesel) and two crude oils to three species of shrimps and mysids – the two fuel oils were more toxic than the crude oils, and the heavy fuel oil was more toxic than the lighter distillate (Anderson et al. 1974). Within Australia, crude oils from different oil fields show a range in density (Neff et al. 2000), and so the type and source of oil in an unplanned spill is a very important factor in determining the extent of impact and level of exposure to toxic hydrocarbons. There has been very little work done specifically on the toxicity of natural gas condensates which are particularly relevant to Australia's NWS but these are known to show toxicity to coral larvae and to affect coral reproduction.

Apart from their differing toxicities, different oil types behave differently in a spill according to their density. Edgar et al. (2003) attributed the minimal impacts of the 2000 *Jessica* spill in the Galapagos, in part, to the thinning effect that the diesel fuel had on the heavy bunker fuel when the two mixed following the spill. In that spill, other circumstances also mitigated the impacts of the spill (e.g. waves, evaporation and currents moving the oil offshore).

Weathering of oil is the process of evaporation of some of the volatile fractions from floating spilled oil and its dilution, modification and breakup by wave mixing, UV radiation, chemical reactions and biological degradation. Oil that has time to weather to a significant degree before it reaches and influences intertidal or benthic habitat will have less toxic impacts that freshly spilled oil. Apart from the oil's composition (or type), the main factors that will influence weathering are temperature and wind speed. Chemical dispersants are designed to accelerate the weathering process.

Although not conducted on Australian invertebrate species, Neff et al. (2000) compared the toxicities of light (Wonnich, Campbell) and medium density (Agincourt) north-west Australian crude oils, and Australian diesel oil, on penaeid prawns, mysid shrimps and sea urchin larvae. They used the oils and several dilutions of pre-distilled WAFs to simulate weathering and assess and compare toxicity. In general, weathering significantly reduced the toxicity of all oils, with variable toxicity between test animals and when different oils were applied (although weathering had a minimal effect on changing the toxicity of the oil to the crustaceans). The heavier Agincourt crude had minimal toxicity on all but the prawns, while for Wonnich, Campbell and diesel oils the percent of the WAF that resulted in LC50 after 96 hours (determined by regression) ranged from 30–48% WAF for prawns and mysids. For sea urchin larvae, results were expressed as percent of the WAF that resulted in abnormal development on 50% of the larvae after 60 hours (60 hour EC50). These varied between 11% and 68% of the WAF for Wonnich and Campbell oils, depending on the type of sea urchin larvae. For diesel, the 60 hour EC50 varied from 27% to 100% (non-toxic) depending on the type of sea urchin larvae used.

3.3.3 Use of dispersants and shoreline clean-up

Debate exists over the merit of using dispersants to help break up oil spills and mitigate the impacts of oil toxicity, as opposed to allowing weathering and natural break-up of the oil slicks or other methods such as burning. Although this is not the only consideration in the use of dispersants, in general most studies have concluded that dispersed oils are more toxic than the oil on its own. Southward and Southward (1978) determined that most of the ecological damage caused by the 1967 *Torrey Canyon* spill in England was due to the use of dispersants and other cleaning measures. Almost all studies we examined found that in a WAF, dispersed oil is more toxic than the oil alone. For example Fisher & Foss (1993) determined that a dispersed oil water fraction using two commercial oil dispersants (Corexit 7664 and Corexit 9527) was ten times more toxic to embryos of grass shrimp (*Palaemonetes pugio*) than the oil water fraction on its own (Fisher & Foss 1993). More recently, Almeda et al. (2013) found that dispersed oil was more than three times more toxic to mesozooplankton than crude oil alone (Almeda et al. 2013). In further experiments, Almeda et al. (2014b) compared growth and survival of nauplii larvae of the barnacle *Amphibalanus improvisus* and tornaria larvae of the enteropneust (acorn worm) *Schizocardium* sp. when exposed to crude oil and dispersed crude oil (using Corexit 9500A). They found that the dispersed oil had a greater toxicity (Almeda et al. 2014b) and they reached the same conclusion when studying the same impacts on the copepods *Acartia tonsa*, *Temora turbinata* and *Parvocalanus crassirostris* (Almeda et al. 2014a). They concluded that the application of dispersants was likely to have a greater effect on post-spill recruitment of marine invertebrates than crude oil alone. The widely consistent demonstration that dispersed oils are more toxic to marine invertebrate than water-oil mixtures alone has led to calls for oil spill cleanups to employ burning off the volatile fraction instead of using dispersants (Georgiades et al. 2003). Although the increased toxicity of dispersed oils has been demonstrated in numerous studies, the practice may be warranted in situations offshore, where reduced shoreline oiling would result from using dispersants despite the higher toxicity to marine life in deeper offshore waters. This consideration is particularly relevant in the Australian context, for oil spills that occur far offshore, such as the Montara wellhead blowout.

3.3.4 Importance of water temperature and exposure time

Because temperature influences the dissolved fraction of oil in water, higher water temperatures dramatically affect the toxicity of both monocyclic aromatic hydrocarbons (MAHs) and PAHs. This needs to be taken into account when conducting experiments and applying laboratory results to the real world, as impacts may vary in temperate and tropical regions and between seasons. Jiang et al. (2012) studied the effect of temperature and exposure time of several zooplankton species to a WAF using crude oil from China. Regardless of temperature, increasing exposure time from 24 hours to 72 hours generally doubled the toxicity (e.g. halved the LC50 concentration). Increasing the temperature from 8.5^{o}C to 16.5^{o}C, and then to 31.2^{o}C, also doubled the toxicity at each temperature step. For example, for the copepod *Labidocera euchaeta*, 24 hour LC50 concentrations changed from ~22 mg/L to 13 mg/L to 4 mg/L at 8.5^o, 16.5 and 31.2^{o}C, respectively. Consistent with this, Fisher and Foss (1993) compared toxicity of fuel oil to embryos of grass shrimp *Palaemonetes pugio* at different temperatures, and found that the effects of toxicity onset earlier at higher temperatures.

3.3.5 Depth, wave exposure and habitat influences

Intertidal and shallow subtidal habitats

Our assessment of the literature confirms some general patterns. Typically, communities on exposed rocky shores are less impacted by oil spills than on sheltered shores, and intertidal communities on the higher part of the shore are usually more impacted than those on the lower shore and subtidal communities. Salt marsh, mangrove and other intertidal sedimentary habitats are probably no more vulnerable to the initial impacts from oil spills than sheltered intertidal rocky shores, but they recover much more slowly due to the residual oil effects caused by sedimentation, re-supply of disturbed sediments (see Section 3.3.1), and loss of perennial habitat-forming macrophytes.

Thus subtidal communities on exposed rocky coasts subject to oil spills (as sometimes occur from shipping accidents away from ports), generally escape significant impacts. Examples of this experience are described by (Edgar & Barrett 2000, Lougheed et al. 2002, Edgar et al. 2003) for shipping accidents in Australia and the Galapagos. Nevertheless exceptions exists, such as when quantities of oil released are very great (e.g. the 2002 *Prestige* oil spill in Spain) and/or when the oil discharges continue for many months (e.g. the 1957 *Tampico Maru* in Mexico; North et al. 1965; Mitchell et al. 1970).

For intertidal rocky shores, Sell et al. (1995) reviewed case studies of 21 spills and showed that exposed rocky shores recovered more quickly than sheltered or moderately exposed shores. This pattern is consistent with the general patterns described above, and is reflected in a number of studies. Lopes et al. (1997) studied the impact of a burst oil pipeline spill in Brazil, where such accidents are common (171 spills between 1974 and 1994). Oiled areas of rocky intertidal shore resulted in immediate mortality of crabs, littorinid snails and isopods, while areas monitored adjacent to the impacted area showed no significant change in cover of barnacles (*Chthamalus* and *Tetraclita*) and mussels (*Brachidontes*). Although doubt exists over the extent of actual oiling of the area surveyed (the impacted area was chosen because of extensive before impact data, with the authors noting that adjacent areas were more heavily oiled), the study indicates a lower level of sensitivity of barnacles and mussels to oil spills on rocky shores where wave action can rapidly disperse, dilute and naturally remove oil. Kotta et al. (2008) also found sheltered and deeper sites were more impacted by oiling than shallow, exposed sites following a large 2006 spill in the Baltic Sea off Estonia. On Macquarie Island, invertebrate abundances at impacted sites on exposed shores had returned to pre oil spill impact levels and were comparable to control sites after seven years, but that this was not the case among the more sheltered kelp holdfast habitat, which had not recovered after that time (Smith & Simpson 1995).

A number of studies of oil spill impacts on intertidal rocky shores have found differential mortality of invertebrates at different positions of tidal height on the shore. This difference is often expressed as different types of animals being more or less sensitive. However our view from this review is that, while taxa clearly differ to some degree in sensitivity to oiling (see Section 3.1.2), observed differences in the field following oil spills are generally due more to zonation of intertidal animals rather than differential sensitivities. In particular, regardless of taxa, invertebrates in the lower intertidal zone of rocky shores generally suffer lower mortality than those in the upper intertidal zone. For example, mussels and barnacles escaped mortality due to their position lower in the intertidal zone following the 1978 *Amoco Cadiz* spill in France, while gastropods higher on the shore were killed (Chassé 1978). However, where heavy oiling of mussels has occurred (e.g. the 1957 *Tampico Maru* spill in Mexico), they too suffered high mortality (Mitchell et al. 1970). Similarly after the 2002 *Prestige* spill in Spain, the extent of mortality of barnacles (often regarded as being resistant to the effects of oil) was found to be depend on the extent of oiling at different locations (Penela-Arenaz et al. 2009). In this case, the barnacle involved was *Chthamalus montagui*, which occupies the higher intertidal area (Penela-Arenaz et al. 2009). Thus, their observations support our assessment that tidal height is more important than species sensitivities, especially in spills involving heavily oiling.

Marine invertebrates occupying intertidal sedimentary habitats such as mangroves, salt marshes, mud flats, sand flats and beaches are especially vulnerable to oil spills. For example, massive mortality of 14.5 million bivalves of the families Cardiidae, Solenidae, Macridae and Veneridae occurred after the *Amoco Cadiz* spill in France (Conan 1982), while Spanish beaches affected by the 2002 Prestige spill initially lost up to 67% of species richness (de la Huz et al. 2005). Similarly, elsewhere in this review we discuss direct mortality of intertidal amphipods, isopods and burrowing crabs by oil spills, and the long recovery times in intertidal habitats. Perhaps the most extreme examples are impacts following the 1969 *Florida* barge oil spill in West Falmouth, Massachusetts, USA (Krebs 1977, Krebs & Burns 1977), and the 1986 refinery spill at Galeta, Panama (Burns et al. 1993). Sedimentation of oil into intertidal habits prolongs the exposure of animals in these habitats to residual oil, both through burrow activities and as the oil is re-exposed due to erosion or other disturbances (see Section 3.3.1). Nevertheless, some evidence suggests that invertebrates on intertidal beach habitats respond differently to those on intertidal rocky shores with respect to zone of greatest impact. Following the 2009 *Pacific Adventurer* oil spill in Australia, Schlacher et al. (2010) found greater impacts lower on the shore on beaches, rather than on the upper shore. For the same spill, impacts where greater on the high intertidal part of the rocky shore than in the lower intertidal (Stevens et al. 2012).

Lastly, where subtidal sedimentary and seagrass habitats are affected by oil spills, high mortalities can also occur. For example, subtidal heart urchins and amphipods were decimated by the spill from the *Amoco Cadiz* that sank 5 km offshore from the Brittany coast of France (Conan 1982). The significance of the effect of oil on subtidal sedimentary habitats is likely to vary according to the size of the spill, depth and degree of mixing, and interactive effects involving exposure. Penela-Arenaz et al. (2009) reported that the heart urchin *Echinocardium cordatum* in Spain was not affected by the *Prestige* oil spill. The sheer scale of the Ixtoc I wellhead blowout in the Gulf of Mexico (475,000 tonnes, of which 120,000 tonnes sank to the bottom) over 290 days during 1979 and 1980 is thought to have resulted in significant subtidal impacts along the Gulf's sandy shores, where it is estimated that all crabs present along 100's of km of coastline were killed (Jernelöv & Linden 1981). Post-spill monitoring of subtidal seagrass beds (> 1 m deep) following the 1986 refinery spill at Galeta, Panama (Jackson et al. 1989) indicated that, relative to control areas, amphipods, tanaidaceans, crabs and ophiuroids were severely impacted by the spill, with very slow recovery, while bivalves, gastropods and polychaetes were either less impacted or recovered over 18 months.

Deepwater habitats

Few studies inform us about the impact and/or recovery of oil spills on deep benthic habitats and their fauna. Guidetti et al. (2000) compared fauna in impacted and non impacted subtidal areas in 75–80 m of water eight years after the 1999 Haven oil spill in the north-west Mediterranean Sea off Italy. Tar aggregates remained in the impacted areas, but comparison of numbers of macroinvertebrates (including polychaetes, sipunculids, bivalves and crustaceans – tanaidaceans, isopods and amphipods) between areas with and without tar aggregates showed no significant differences. Following the 2010 *Deep Water Horizon* blowout in the Gulf of Mexico, Felder et al. (2014) found that shrimp, crab and lobster species associated with rhodolith and macroalgal habitat in 55–80 m water depth decreased dramatically in both diversity and abundance. They concluded this was an indirect effect of the oil killing the algae and rhodoliths, rather than actual mortality from the oil. Montagna et al. (2013) found that in very deep water (~1200 m), copepods were affected negatively by the same spill although nematodes responded positively (probably due to nutrients enrichment after the spill and subsequent increased bacterial activity). After the 2002 *Prestige* oil spill off Spain, Sanchez et al. (2006) surveyed depths of 70–500 m where tar aggregates from 200–300 kg/km^2 existed over an area ~25 km in diameter. This area had good historical data from fisheries surveys, which was used to identify reductions in the abundance of the Norwegian lobster *Plesionika heterocarpus,* which were attributed to direct mortality from the oil spill.

4. Capacity for recovery of marine invertebrates

4.1 Evidence for short term recovery (months to a few years)

Sell et al. (1995) reviewed studies on 34 oil spills to form a generalised view that times taken for community assemblages to recover after an oil spill are about 3 years for rocky shores and five years for salt marshes. However numerous exceptions were identified. Indeed very few of the case studies were carried through to full "*recovery*" (in this case defined as "*where a natural biota has been established and is within the range of dominance, diversity, abundance and zonation expected for that habitat*"). Sell et al. (1995) found that in most cases recovery on heavily oiled, exposed rocky shores is well advanced by 2 years and in the one case available, fully recovered within three years, except where the shoreline had been subject to mechanical and/or chemical cleaning treatment. In such cases, recovery was not complete after four years in the studies examined. For shores that are heavily oiled, moderately exposed and sheltered, few case studies are available where post-spill cleanup had not been carried out. Mostly these sites were in a recovering stage after two to three years, except in three cases where they were still in the recovery phase after ten years. Sell et al. (1995) noted that in at least one of these cases, the cleanup procedure was particularly intense. For moderate to lightly oiled rocky shores, recovery was much more rapid, that is within one to five years (mostly two), except where shoreline clean up had been undertaken. For heavily oiled salt marsh habitat, Sell et al. (1995) found periods of three to six years for full recovery (with some exceptions); treated shorelines took longer to recover.

The review by Sell (1995) provides an excellent “big picture” assessment, although after 20 years is overdue for an update. However, results are simplified because their approach does not discriminate important differences in spill circumstances, particularly gradients of impact and differential impacts on different taxa within the same spill.

Eighteen months after a large oil spill off Estonia in 2006, Kotta et al. (2008) found that herbivores (especially amphipods and isopods) had not recovered, while deposit feeders and suspension feeders (which had not been impacted) remained stable. Elmgren et al. (1983) found that three years after the oil *Tsesis* oil spill, *Pontoporeia* amphipod numbers were still depressed, while the polychaete *Harmothoe sarsi* had returned to pre-impact densities and *Macoma balthica* (which had not been affected by the spill) became significantly more abundant. They estimated it would take five–ten years (or even longer) for the relative densities of species in the affected area to return to normal. Garrity & Levings (1990) found significant impacts on mollusc populations on intertidal mudflats in Galeta, Panama after a 1986 refinery spill. They estimated it would take five to ten years (or perhaps longer) for the relative densities of species in the affected area to return to normal. Garrity and Levings (1990) Garrity & Levings (1990) found significant impacts on intertidal mudflat populations of molluscs in Galeta, Panama, after a 1986 refinery spill. They monitored impact and control sites, and after three years recruitment in the spill areas remained reduced relative to un-impacted sites; some species of littorinids had not re-established after three years. After the 1999 *Erika* spill in France resulted in 100% mortality of the sea urchins *Paracentrotus lividus* and *Psammechinus miliaris*, Barillé-Boyer (2004) found it took two years until the first urchins reappeared and three years for densities to return to pre-impact levels (~60 per m^2).

In general, studies of simulated oil spills have demonstrated rapid recovery from recolonisation from adjacent plots (McGuinness 1990, Egres et al. 2012), but this is likely to be unrealistic in an extensive real world spill. However a large-scale (900 m^2) simulated oil spill in Panama caused total mortality of the sea urchins present, and those that re-appeared seven months later (as recovery started to take place) were smaller and most likely recruits rather than migrants (Ballou et al. 1989).

4.2 Evidence for long term recovery (> 5 years)

Despite some studies finding good recovery after periods of 3–6 years, others have demonstrated incomplete recovery or lagging sublethal effects that persisted for decades. Incomplete recovery (to the extent it should be regarded as permanent impact) has occurred in some of the worst examples of oil spills, although this point is hotly debated as in the case of the *Exxon Valdez* spill (Peterson et al. 2003, Harwell & Gentile 2006, Payne et al. 2008, Bodkin et al. 2014).

Studies reviewed here suggest that in intertidal sedimentary habitats such as salt marshes and mangroves, effects of oiling can last decades. Krebs and Burns (1977) found significant effects of the *Florida* barge oil spill, which affected salt marsh habitat in West Falmouth, Massachusetts, USA, remained after 7 years with fiddler crabs (*Uca punax*) at reduced densities compared to pre-spill levels and residual chronic effects on crab health and behaviour. Twenty years after the same spill, Teal et al. (1992) found that sedimented oil still remained at 10–15 cm below the surface in the heaviest oiled areas in sufficient levels to affect crab utilisation of habitat. They concluded that if these sediments were disturbed such that the oil was again exposed at the surface, it would lead to toxic concentrations of oil re-occurring. Another survey 30 years after the event (Reddy et al. 2002) found similar results, with oil still present in cores between 6 and 28 cm from the surface. Carls et al. (2001) measured the rate of decline in hydrocarbons in mussel beds and the underlying sediment affected by the 1989 *Exxon Valdez* oil spill, and concluded that it would take another 30 years to recover. Harwell and Gentile (2006) reviewed studies of the impacts of the *Exxon Valdez* spill, and concluded that with the exception of killer whales, other species (including the invertebrates) that had suffered a decline in abundance following the spill had recovered in abundance within 6 years. However, Peterson (2001) and Peterson et al. (2003) concluded that persistent low-level exposure to residual sedimented oil from the *Exxon Valdez* continued to cause impacts to several species, with sublethal effects likely to continue for many years as continued exposure from oil in contaminated sediments resulted in bioaccumulation of hydrocarbons by macroinvertebrates and then through the food chain to birds and mammals.

Beyond the issue of sublethal effects, numerous examples in the literature indicate partial recovery within a few years, as suggested by Sell et al. (1995), but with long term effects persisting for many more years. The 1957 *Tampico Maru* wreck in Baja California, Mexico partially blocked a cove and spilled oil for 8–9 months. Despite this incident, the intertidal gastropod *Littorina planaxis* survived, but subtidal gastropods including abalone (*Haliotis* spp.) were killed, with reduced numbers 7 years later (North et al. 1965). Conan (1982) noted that after the 1978 *Amoco Cadiz* spill of 223,000 tonnes of crude oil in France, delayed effects on mortality, growth and recruitment were still observed up to three years after the spill. This author estimated it would take 3–6 generations (5–10 years in the case of bivalves) before populations recovered fully. After the 2002 *Prestige* oil spill in France, Castege et al. (2014) showed that despite the recovery of many taxa, nine years after the spill the sponges *Hymeniacidon perlevis* and *Tethya* sp., which had dominated some parts of the lower intertidal area before the spill, had not returned.

Egres et al. (2012) experimentally oiled sandy intertidal sand flat plots with diesel in Brazil in order to simulate a small spill of fuel oil. They documented high mortalities of animals (e.g. gastropods, oligochaetes and ostracods) in the plots (just 0.35 m^2, each oiled with 2.5 L) and then rapid recovery with two days from recolonisation from outside the impacted plots. While studies such as this reinforce the toxic nature of hydrocarbons in the field, they do not greatly inform determination of recovery rates except to say that where oiling is not heavy enough and persistent enough to be incorporated into sedimentary processes, recolonisation of sand habitats can occur rapidly.

4.3 Assessing impact and recovery from oil spills

A number of key lessons learned are apparent from the literature when assessing the impacts of oil spills on marine invertebrates, as well as other animals and plants.

- Best practice is a Before-After-Control-Impact with well established time series of baseline data (Green 2005). Good examples in the Australian context are Edgar & Barrett (2000) and MacFarlane & Burchett (2003), although often the availability of pre-impact baseline is fortuitous, having been established for other purposes. The lack of baseline data at affected and unaffected sites prior to an oil spill is regarded as the greatest impediment to understanding the effects of oil spills (Bodkin et al. 2014). Some of the most informative studies were where multiple years of pre-spill baseline data existed in impact and un-impacted areas. Most often this was fortuitous, such as the overlap of high human population density, regional research laboratories, and shipping lanes that exists in Europe. Australia's dispersed population, extensive remote coastlines and highly centralised research infrastructure is unlikely to see such inadvertent preparedness (but see Edgar & Barrett 2000; MacFarlane & Burchett 2003). A better planned approach is to establish baselines in areas of intensive or frequent industrial use, as well as control sites, as occurred with the long term seagrass monitoring program in Gladstone, which was established several years before the 2006 *Global Peace* spill in Gladstone and subsequently provided an excellent pre-spill baseline (Taylor & Rasheed 2011). In the absence of a baseline, a "bullseye" sampling design that establishes a gradient between impact and control is recommended (Green 2005). In the absence of baseline data ahead of an oil spill, Smith and Simpson (1995; 1998) and Schlacher et al. (2010) have demonstrated it is possible to make very good control-impact only comparisons by repeated measure studies of oiled and non-oiled locations.

- Sampling design with sufficient power to detect change needs to be used. This would seem to be self evident; however Peterson et al. (2001) showed that some of the assessments undertaken following the *Exxon Valdez* spill in Alaska had insufficient power to detect an impact where it had occurred, and that this had contributed to the controversy over the severity of spill impacts (Peterson et al. 2001).

  Impacts should be assessed outside the zone of maximum damage. This is likely to be the high impact zone, but it may be small in area relative to the areas of more moderate or lesser impact. Assessing these areas, in addition to the main impact sites and control sites, is likely to be more informative about gradients of impact and recovery times, relative sensitivities of different species and sublethal impacts.

- Indirect and sublethal impacts are important. Numerous studies have pointed to the ongoing long term sublethal effects of oil spills (e.g. (Johnson 1977, Peterson 2001); see Section 3.2), and interactions between exposure to oiling, environmental conditions, habitat, biota and behaviour.

- Monitoring should continue until recovery is complete or change has stabilised. Very few studies have followed the course of recovery from an oil spill through to full recovery. Most ceased while the communities were in a recovery mode or well on the way to recovery (Sell et al. 1995). Researchers occasionally returned after 20 even 30 years, but generally these were in the worst cases where recovery may never be complete or permanent (e.g. (Teal et al. 1992, Reddy et al. 2002) in the case of the 1969 West Falmouth, USA spill). Intervals between monitoring surveys can increase with time.

- Archive samples for future analysis. Changes in analytical methods, improved detection limits and technological improvements in instrumentation over the course of a long running monitoring program (such as occurred with the 1989 *Exxon Valdez* spill; (Payne et al. 2008)) can confuse interpretation and contribute to debate about impact and recovery.

5. Conclusions

Marine invertebrates are sensitive to the toxic effects of oil. Depending on the intensity, duration and circumstances of the exposure, they can suffer high initial mortality together with prolonged sublethal effects, which can act at both the individual and population level. Under some circumstances, recovery from these impacts can take years to decades. However the key factor associated with impact is exposure, with many factors mitigating the degree of exposure, meaning that in many cases impacts are moderate and recovery occurs within a few years. While a range of generalities can be stated about the response of marine invertebrates to oil spills, almost every oil spill is unique in terms of its impact because of differing levels of exposure. The variety of factors that contribute to exposure include: type of oil, amount of oil, extent of weathering, whether the exposure is transient or persistent, whether dispersants or other cleanup measures were used, the type of habitats and depths affected, the species present and their stage of development or maturity, the species assemblages present, and how the process of recolonisation proceeds in terms of recruitment and other dynamics. The importance of each of the factors and how they affect the degree of impact have been explored in this review.

The type and source of oil in an unplanned spill is a very important factor in determining the extent of the impact and the level of exposure to toxic hydrocarbons. Refined oils, diesel and heavy bunker fuel oils are more toxic than crude oils and the type of oils spilled usually depends on whether the accident involves a cargo ship, an oil tanker, a refinery spill or a wellhead blow-out. In Australia crude oils from different oil fields possess a wide range of densities.

The degree of exposure to marine invertebrates from an oil spill will depend in part on the degree of weathering and dilution (e.g. from wave action) that takes place from the time oil makes contact with the shoreline or benthos. In general, the part of weathering that involves evaporation will drive off some toxicants but it also serves to concentrate others and affects the consistency of the oil. In terms of toxicity to marine invertebrates, it is the extent of dilution that determines exposure concentrations and is more important than a short term period of weathering. Almost without exception among the studies we reviewed, the use of dispersants in the nearshore environment and mechanical cleaning of oil from the substrate resulted in higher mortality and longer recovery times than the oil on its own. Dispersed oils are significantly more toxic than oil alone, but their use offshore to improve dilution, aid weathering and decrease concentrations before shorelines are impacted may be warranted.

The type of habitat impacted by the oil, including substratum type, depth and wave exposure, is a very significant factor in modulating impacts on marine invertebrates in those habitats. Typically, communities on exposed rocky shores are less impacted by oil spills than on sheltered shores, and intertidal communities on the higher part of the shore are usually more impacted than those on the lower shore and subtidal communities. Salt marsh, mangrove and other intertidal sedimentary habitats are probably no more vulnerable to the initial impacts from oil spills than sheltered intertidal rocky shores, but they recover much more slowly due to the residual oil effects caused by sedimentation and re-supply as sediments are disturbed.

Thus oil spills that result in significant exposure to intertidal sedimentary habitats cause high levels of mortality to marine invertebrates, with prolonged chronic effects when oil is incorporated into sedimentary layers, which causes disruption to burrowing and bioturbation. Oil is easily sedimented into these types of habitats, with resultant risk of re-exposure of oil through erosion or by animals foraging and burrowing. Several studies reviewed in this white paper showed that oil incorporated in sediments can release lethally toxic levels of hydrocarbons at least two decades after the initial exposure. Loss of salt marsh and mangrove habitat resulting from oil spills in sedimentary habitats also impacts marine invertebrates – first directly from exposure and loss of habitat, and later from increased predation (e.g. from birds) as the plant cover remains thinned for some time. In sedimentary habitats, the taxa predominantly impacted (at least in the studies reviewed) were small crustaceans such as amphipods, crabs, bivalves and gastropods. However some taxa like cerithiid gastropods (creepers) and some nematodes and polychaetes seem resilient to oiling. For subtidal sedimentary habitats and seagrass beds, amphipods, crabs, bivalves, gastropods and sea urchins are affected but unless the spill is particularly heavy (as has occurred in some spills in Europe and Panama), subtidal habitats are less frequently impacted than intertidal habitats. In Australia, spills have impacted mangrove and beach habitats, resulting primarily in mortality to crustaceans and gastropod molluscs.

For intertidal reef or rocky shore habitats, the initial contact from heavy oiling causes high mortality, especially of small crustaceans like amphipods and isopods, gastropod molluscs and echinoderms. The few studies that have assessed impacts on sponges suggest they are particularly sensitive to the impacts of oil and slow to recover. In tropical areas, scleractinian and hydrozoan corals are affected along with anemones and zoanthids. Longer term effects of exposure to oil appear to be less on exposed rocky intertidal habitats than on those that are sheltered, apparently because exposed intertidal habitats are vigorously washed by turbulent waves and oil is less likely to be trapped within the substratum. Overall, with some caveats, intertidal rocky shores appear more resilient to the long term effects of oil spills but not to initial exposure mortalities. Subtidal reef habitats also suffer mortalities, but these are less than on intertidal reefs and limited to very heavy spills of oil from ships or refineries close to shore. There have been no significant impacts to subtidal reefs detected from oil spills in Australia.

Several studies claim their results show that barnacles and mussels are less sensitive to oiling than other taxa. We found little evidence for this, and suggest these findings more likely reflect differences in exposure and that impacts of oiling on rocky intertidal shores are consistently more severe in the high intertidal zone compared with the lower intertidal. Intertidal herbivorous gastropods, small crustaceans and sea urchins appear particularly vulnerable to oil spills, with high mortalities recorded in many studies. It is possible that this reflects the abundance of this type of animal in intertidal habitats relative to other taxa (including predatory gastropods, for example), which may have been initially uncommon enough to allow statistically valid comparisons to be made between oiled and control sites. Such species often exhibit low densities and high variance-to-mean ratios, making comparisons between sites difficult. Thus, it is possible that a broader range of invertebrates than those commonly studied to determine impact response might also be just as vulnerable to oil spills. Crabs are another group that appear particularly vulnerable based on the studies reviewed. This was particularly true of burrowing species in sedimentary habitats.

The indirect effects of oil spill impacts on marine invertebrates include changes to dominance patterns in community assemblages. In the studies we reviewed, two types in particular were evident. Firstly, on soft sediment habitats some species of nematodes and polychaetes dominated recovery processes and achieved very high abundances relative to pre spill levels. This was due to either (or both) their relative insensitivity to the oil and/or their quickness to recolonise, including in response to the organic enrichment that comes with oil sedimentation. Release from competition when formerly dominant taxa are removed by oiling may also contribute. Secondly, on rocky intertidal shores and subtidal reefs, heavy mortality of grazing amphipods, gastropods and urchins was followed by a proliferation of opportunistic algae taking advantage of the lack of grazers, the space cleared by the death of sessile invertebrates, and organic enrichment.

However the key factor in all these considerations of oil impacts on marine invertebrates is the level of exposure. Where oiling is slight because of low concentration exposure (as might occur far from the spill site or in a small spill), the impacts of oil spills on marine invertebrates appear low or at least short lived. That said, sublethal impacts (e.g. impaired motility in crabs, lower adhesion strength in limpets) have been found at very low concentrations, and these may be sufficient to cause the animals to be unable to feed or avoid predators, thus affecting their likelihood of survival. Similarly, low concentrations of hydrocarbons can result in a range of other sublethal effects such as reproductive impairment that causes effects at the population level (reproductive and/or recruitment failure, disease, DNA damage and loss of genetic diversity). Bioaccumulation of hydrocarbons in crustaceans and in bivalve molluscs impacts reproductive success and results in transfer of hydrocarbons higher up the food chain. In general, oil appears more highly toxic to larval invertebrates than adults, so this impact needs to be considered.

In the Australian context at least, the level of exposure to oil and subsequent impact will differ greatly between offshore wellhead accidents (e.g. *Montara* 2009), shipping accidents close to shore (e.g. *Pacific Adventurer* 2009), and refinery/oil storage depot spills (e.g. Port Stanvac refinery 1999), and pre-spill planning and baseline assessment needs to be considered differently. Even though the volumes of oil likely to be involved are much greater from wellhead blowouts, the risk of direct impacts, at least to intertidal and shallow subtidal reefs and sedimentary habitats, seems low in comparison to other types of spills. For this reason, pre-spill precautionary assessments should not just seek to establish baselines against which to assess impact, but should determine the risk to exposure of a range of oil-water fractions and hydrocarbon concentrations of marine invertebrates and to test the response of a range of marine invertebrate receptors to those concentrations. Both lethal and sublethal responses need to be assessed, and perhaps most importantly given the differing toxicities of different types of crude and refined oils, the assessments need to be done using the oil with highest risk in terms of local geography.

In this review, we examined the records of assessment of 44 significant oil spills in Australia since 1970. Of these, five occurred offshore with no likely or expected impact on benthic invertebrates. Despite the potential for oil spills to impact marine invertebrates, only 21 cases had potential direct effects of oil studied, and 18 cases had only cursory or no assessment of impact. Of those

21 spills where impact assessments are available in published or unpublished reports, only 8 considered impacts on invertebrates, with many others focusing on the primary plant habitat affected but with little or no consideration of the invertebrate communities they support. With the exception of the 2009 *Montara* wellhead spill and the 1999 *Torungen* spill, detailed assessment of spill impacts on invertebrates in Australia have been limited to temperate waters.

We also found very few assessments of the toxicity and sublethal effects of oil on Australian marine invertebrates. Those that have been undertaken have been in south-eastern Australia. While they are useful in the local context, a high priority remains to test the responses of Australian marine invertebrates across the range of habitats and geography and types of oil they might be exposed to. Given the nutrient deficient status of Australian seas relative to the North American and European locations where most studies have been undertaken, the concentrations of oil needed for lethal impacts on invertebrates may well be much lower in Australia. In particular, we identified a number of taxa of habitat forming, sessile, filter feeding invertebrates (sponges, bryozoa, tunicates) that need assessment of their response to oil. In addition, more information is needed about Australian species in different parts of Australia for taxonomic groups that are known from overseas studies to be vulnerable to exposure to oil – these include molluscs, crustaceans and echinoderms. Lastly, in this review we outlined some of the lessons learned in assessment of oil spill impacts from the studies examined, and provided some recommendations to be considered in responding to oil spills in Australia.

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