

Oceanography and Marine Biology

An Annual Review

VOLUME 56

S.J. Hawkins, A.J. Evans, A.C. Dale,
L.B. Firth & I.P. Smith

First Published 2018

ISBN 978-1-138-31862-5 (hbk)

ISBN 978-0-429-45445-5 (ebk)

Chapter 5

Impacts and Environmental Risks of Oil Spills on Marine Invertebrates, Algae and Seagrass: A Global Review from an Australian Perspective

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CRC Press

Taylor & Francis Group
Boca Raton London New York

CRC Press is an imprint of the
Taylor & Francis Group, an **informa** business

IMPACTS AND ENVIRONMENTAL RISKS OF OIL SPILLS ON MARINE INVERTEBRATES, ALGAE AND SEAGRASS: A GLOBAL REVIEW FROM AN AUSTRALIAN PERSPECTIVE

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Abstract

Marine invertebrates and macrophytes are sensitive to the toxic effects of oil. Depending on the intensity, duration and circumstances of the exposure, they can suffer high levels of initial mortality together with prolonged sublethal effects that can act at individual, population and community levels. Under some circumstances, recovery from these impacts can take years to decades. However, effects are variable because some taxa are less sensitive than others, and many factors can mitigate the degree of exposure, meaning that impacts are moderate in many cases, and recovery occurs within a few years. Exposure is affected by a myriad of factors including: type and amount of oil, extent of weathering, persistence of exposure, application of dispersants or other clean-up measures, habitat type, temperature and depth, species present and their stage of development or maturity, and processes of recolonisation, particularly recruitment. Almost every oil spill is unique in terms of its impact because of differing levels of exposure and the type of habitats, communities and species assemblages in the receiving environment. Between 1970 and February 2017, there were 51 significant oil spills in Australia. Five occurred offshore with negligible likely or expected impacts. Of the others, only 24 of the spills were studied in detail, while 19 had only cursory or no assessment despite the potential for oil spills to impact the marine environment. The majority were limited to temperate waters, although 10 of the 14 spills since 2000 were in tropical coastal or offshore areas, seven were in north Queensland in areas close to the Great Barrier Reef. All four spills that have occurred from offshore petroleum industry infrastructure have occurred since 2009. In Australia, as elsewhere, a prespill need exists to assess the risk of a spill, establish environmental baselines, determine the likely exposure of the receiving environment, and test the toxicity of the oil against key animal and plant species in the area of potential impact. Subsequent to any spill, the baseline provides a reference for targeted impact monitoring.

Introduction

Background to this review

The aim of this paper is to review the impacts of oil and oil spills on marine invertebrates, seagrasses and macroalgae, particularly in an Australian context. Reviewing oil spill literature is a daunting task, and there is always the risk of omission. For example, Harwell & 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). For marine algae and seagrasses, a detailed review was provided by O'Brien & Dixon (1976), while the toxicity of oil to aquatic plants was recently reviewed by Lewis & Pryor (2013). Apart from a report by Runcie et al. (2005) on seagrasses, no detailed review exists on the impacts of oil on marine flora and fauna from an Australian perspective.

Our assessment fully evaluates the small number of Australian studies that we are aware of and captures both the primary and grey literature. We then 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 to identify the largest gaps in our ability to infer likely impacts of any future spills in Australia.

The scope for our review includes marine macrophytes and all life stages of marine invertebrates with a benthic adult phase. We also include zooplankton, but we do not consider impacts on phytoplankton or microbial communities. Other studies review and experimentally evaluate how microbial communities respond to oil (MacNaughton et al. 1999, Nyman & Green 2015); the review by Lewis & Pryor (2013) covers microalgae to some degree. 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, ratios in oil–water mixes and methods of creating water-accommodated (soluble) fractions (WAFs or WSFs—note we use both terms where different authors used them) (Redman 2015, Redman & Parkerton 2015); consequently, we have not attempted to make such comparisons in reviewing literature in this paper. However, we emphasise that the circumstances and level of exposure to oil are key elements needed when assessing risk, the likelihood of biological impact and impact detection.

The scale and nature of oil spills has changed in recent decades. Previously, major spills were primarily the result of shipping-related incidents, but recent spills from deep-water offshore fixed installations have resulted in markedly different circumstances of exposure of the environment to releases of oil. Over three months from 21 August 2009, a damaged production well (MONTARA) released gas, condensate and ~4750 tonnes of crude oil into the Timor Sea, creating arguably Australia's most significant oil spill. Not long after the MONTARA incident, the DEEPWATER HORIZON oil spill (20 April 2010) occurred in the Gulf of Mexico, with resultant discharge of an estimated 7 million tonnes of crude oil, the largest accidental volume ever released. While both incidents highlighted a general lack of preparedness for such events, the environmental monitoring programmes triggered by these oil spills, especially DEEPWATER HORIZON, resulted in extensive research on the impacts of spills. That both events occurred in warm-water environments is also pertinent to advancing our understanding of the implications of oil spills in an Australian context, since the greatest expansion in oil and gas exploration and extraction has been on the country's tropical north-west shelf. Both these spills provide an opportunity to expand on previous research prompted by major incidents that occurred in temperate/cold-water environments (EXXON VALDEZ and TORREY CANYON by way of examples).

The review is structured as follows. The remainder of this introductory section provides some context of how oil behaves and breaks down over time in seawater and its fate in coastal environments. The second section provides a summary of the marine oil spills of significance that have occurred around Australia since 1970 and then follows a detailed review, structured by geography and/or habitat type, of the impacts of these spills and the studies (if any) undertaken on them. The third section provides a detailed review of the direct and indirect impacts, both acute and chronic, for each of the

major taxonomic groups of invertebrates, seagrasses and macroalgae, and the factors that affect the severity of impact from exposure to oil. The fourth section evaluates the evidence for recovery of species and assemblages from the impact of oil spills, both in the short term and long term, and the factors that affect recovery. This section also sets out some key attributes for good impact assessment based on our review. Finally, we provide a concluding section which summarises the key points from the review and highlights important knowledge gaps that we feel should be addressed.

Behaviour and fate of crude oil in coastal environments

Our review is principally focussed on the interaction between crude oils and marine organisms within intertidal and subtidal marine coastal areas. As such, an important consideration is how the behaviour and fate of crude oils vary in space and time, as this in turn influences the likelihood and type of interaction spill oils may have on these organisms.

Following an oil spill, evaporation will typically remove about one-third of the volume of a medium crude oil slick within the first 24 hours, but a significant residue will always remain (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 (Lee et al. 2015), with the risk of subsequent erosion of oiled sediments from the shoreline and deposition in nearshore habitats. Dissolution from slicks and adsorbed oil can persist for weeks to years (NRC 2003). This is particularly relevant to understanding impacts on benthic invertebrates that principally occupy the seafloor and intertidal zones.

While nearshore oil spills obviously pose a high risk to benthic animals and plants, in an offshore spill, a significant portion of toxic components are likely to be lost before the oil reaches coastal waters or the seafloor (Haapkylä et al. 2007). 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 are entrained into the water column. Dispersant application can dilute oil concentrations to less than 100 ppm under turbulent (1 m) wave conditions (Lee et al. 2013). Within 24 hours of release, it is expected that oil will disperse and mix into the water column and be diluted to concentrations well below 10 ppm, with dilution continuing as time proceeds. As biodegradation and dilution takes place over the following weeks, dispersed oil concentrations could be expected to decline to less than 1 ppm (Lee et al. 2013). The response of organisms exposed to oil maybe acute or chronic. For the purposes of this review, the Australian and New Zealand Environment and Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand (ANZECC & ARMCANZ 2000) definitions are used, whereby acute is defined as a rapid adverse effect (e.g. death) caused by a substance in a living organism. Acute can be used to define either the exposure or the response to an exposure (effect). Chronic is defined as lingering or continuing for a long time, often for periods from several weeks to years. Chronic can also be used to define either the exposure of a species or its response to an exposure (effect). Chronic exposure typically includes a biological response of relatively slow progress and long continuance, often affecting a life stage (ANZECC & ARMCANZ 2000).

Oil spills in Australia and the impact of oil on marine invertebrates, algae and seagrass

Scientific research conducted on Australian oil spills

According to the Australian Maritime Safety Authority (AMSA 2017), there were at least 44 large or notable oil spills in Australia between 1970 and 2012; we have documented a further seven between 2013 and 2017 (Table 1) with the potential to impact marine habitats and the floral and

faunal assemblages occurring in them. Most of these were the result of shipping incidents (42) but also include refinery (1) and bulk storage spills (3), deliberate illegal dumping onshore (1), leaks from platform or seabed oil and gas infrastructure (3) and an oil platform wellhead blowout (1).

Most crude oil and fuel spills in Australian waters have had little or no assessment of possible or realised impacts of oil. Of the 51 spills that have occurred from 1970 to 2017, six were in offshore waters, with no significant impacts (or no expected impact), 24 involved studies of potential effects, and 19 had no or only cursory assessment of impact and two are currently being assessed (Table 2). Much of the work undertaken to date to assess the impacts of oil spills in Australia has not been reported in the peer reviewed literature, thus limiting its utility and accessibility. For example, of the 24 spills cited in Table 2 with published outcomes, documentation on impacts or the assessment of potential impacts in peer reviewed journals occurred in just 10 cases. One of the possible reasons there are limited published studies of oil spill impacts in Australia is that few have been able to be conducted with the foresight, or in some cases fortuity, to have rigorous experimental designs in place; notable exceptions are Edgar & Barrett (2000) and MacFarlane & Burchett (2003). However, several published studies have incorporated appropriate control-impact only comparisons by examining oiled and non-oiled locations, and then repeating measurements for a period of time post impact (e.g. Smith & Simpson 1995, 1998, Schlacher et al. 2011).

Offshore spills and coral reefs

Of the 51 spills listed in Table 1, 10 spills occurred far offshore from the mainland, with only two of these having the potential to cause significant environmental impacts to benthic or intertidal habitats (the 2002

Table 1 Records of significant or notable historical Australian oil spills since 1970

Date	Vessel/source	Location, state	Oil amount (tonnes)
3 March 1970	OCEANIC GRANDEUR	Torres Strait, Qld	1100
1 January 1975	LAKE ILLAWARRA	Hobart, Tas.	81 ^b
4 February 1975	ESSO DEN HAAG	Port Stanvac, SA	<41 ^b
26 May 1974	SYGNA	Newcastle, NSW	700
14 July 1975	PRINCESS ANNE MARIE	Offshore, WA	14,800
24 July 1976	FU LONG II	Geraldton, WA	Unknown ^b
18 December 1976	BETHIOUA	Tamar River, Tas.	356 ^b
8 January 1977	AUSTRALIS	Sydney Cove, NSW	Unknown ^b
7 March 1977	YUN HAI	Newcastle, NSW	100 ^b
31 March 1977	STOLT SHEAF	Ballast Point, NSW	20 ^b
15 March 1978	Unknown	Teewah to Ballina	Unknown ^b
10 September 1979	WORLD	Botany Bay, NSW	95
	ENCOURAGEMENT		
29 January 1981	Unknown	Botany Bay, NSW	50–100 ^c
29 October 1981	ANRO ASIA	Bribie Island, Qld	100
22 January 1982	ESSO GIPPSLAND	Port Stanvac, SA	Unknown
August 1982	Oil barge	Parramatta River, NSW	Unknown ^e
5 February 1985	ARTHUR PHILLIP	Botany Bay, NSW	7–50 ^d
August 1987	Storage tank leak	Withnell Bay, WA	50 ^e
3 December 1987	NELLA DAN	Macquarie Island	125
6 February 1988	SIR ALEXANDER GLEN	Port Walcott, WA	450
20 May 1988	KOREAN STAR	Cape Cuvier, WA	600
28 July 1988	AL QURAIN	Portland, Vic.	184
21 May 1990	ARTHUR PHILLIP	Cape Otway, Vic.	Unknown

Continued

Table 1 (Continued) Records of significant or notable historical Australian oil spills since 1970

Date	Vessel/source	Location, state	Oil amount (tonnes)
14 February 1991	SANKO HARVEST	Esperance, WA	700
21 July 1991	KIRKI	Offshore, WA	17,280
December 1991	Fuel storage depot	Cape Flattery, Qld	8 ^e
30 August 1992	ERA	Port Bonython, SA	300
1993	Unknown	Point Sampson, WA	1 ^e
December 1994	Illegal dumping	Cairns, Qld	6 ^e
10 July 1995	IRON BARON	Hebe Reef, Tas	325
1996	Unknown	Point Sampson, WA	1 ^e
2 March 1998	Fuel storage depot	Christmas Island	60
28 June 1999	Mobil refinery	Port Stanvac, SA	230
26 July 1999	MV TORUNGEN	Varanus Island, WA	25
3 August 1999	LAURA D'AMATO	Sydney, NSW	250
16 December 1999	METAXATA	Ballina, NSW	1400 ^f
18 December 1999	SYLVAN ARROW	Wilson's Promontory, Vic.	<2
2 September 2001	PAX PHOENIX	Holbourne Island, Qld	<1
25 December 2002	PACIFIC QUEST	Border Island, Qld	Unknown
24 January 2006	GLOBAL PEACE	Gladstone, Qld	25
11 March 2009	PACIFIC ADVENTURER	Cape Moreton, Qld	270
21 August 2009	MONTARA Wellhead	NW Australian coast	4750
3 April 2010	SHEN NENG ^a	Great Keppel Island, Qld	4
9 January 2012	MV TYCOON	Christmas Island	102
6 June 2013	FV JOSEPH M ^f	Lady Elliot Island, Qld	<1
29 July 2013	Refuelling accident ^f	Port of Brisbane, Qld	Unknown
7 September 2013	ESSO COBIA platform ^{f,g}	Bass Strait, Victoria, 50 km offshore	<1
3 October 2013	Seafaris tourist catamaran ^f	Cape Tribulation, Qld	Unknown (vessel caught fire and sank with 12–15 t of diesel on board)
17 July 2015	Alleged ship responsible being investigated ^{f,h}	Cape Upstart, Qld	10–15
April 2016	WOODSIDE COSSACK ^d Well ^{i,j}	Pilbara, 150 km offshore Karratha, WA	10.5
1 February 2017	ESSO WEST TUNA platform ^{k,l}	Bass Strait, Victoria, 45 km offshore	Unknown

Source: Unless specified, source for records up to 2012 is Australian Maritime Safety Authority (AMSA). 2017. Major historical incidents. Australian Maritime Safety Authority. Available at <https://www.amsa.gov.au/environment/protecting-our-environment/major-historical-incidents/> (accessed 6 August 2017).

^a Manuell (1979).

^b Dexter (1984).

^c Anink et al. (1985).

^d Duke & Burns (1999).

^e Nelson (2000).

^f AMSA records of pollution events since 2012 (involving oil spills) rated as category 3 (moderate) or above provided by the Australian Maritime Safety Authority.

^g Australian Broadcasting Commission (ABC) (2013).

^h Australian Broadcasting Commission (ABC) (2017a).

ⁱ Australian Broadcasting Commission (ABC) (2017b).

^j *The West Australian* (2017).

^k NOPSEMA (2017).

^l *The Guardian* (2017).

Table 2 Summary of habitats and flora/fauna impacted or potentially impacted by oil spills in Australia since 1970

Year	Vessel/source	Intertidal habitats potentially impacted	Subtidal habitats potentially impacted	Studies on impacts	Description of spill and/or response
Nearshore subantarctic					
1987	NELLA DAN	Rocky shore	Reef	Molluscs, echinoderms, crustaceans: Pople et al. (1990); Simpson et al. (1995); Smith & Simpson (1995, 1998) algae: Pople et al. (1990); Simpson et al. (1995); Smith & Simpson (1995, 1998)	AMSA (2017); Lipscombe (2000)
Nearshore temperate					
1974	SYGNA	Beach	Unknown	None	AMSA (2017)
1975	ESSO DEN HAAG	Unknown	Unknown	None	Manuell (1979)
1976	BETHOUA	Unknown	Unknown	None	Manuell (1979)
1976	FU LONG II	Unknown	Unknown	None	Manuell (1979)
1978	Unknown	Unknown	Unknown	None	Manuell (1979)
1981	ANRO ASIA	Beach	Unknown	None	AMSA (2017)
1981	Not known	Mangroves		Mangroves: Allaway et al. (1985); Allaway (1987) amphipods: Dexter (1984)	Dexter (1984); Allaway et al. (1985); Allaway (1987) AMSA (2017)
1982	ESSO GIPPSLAND	Beach	Unknown, possibly seagrass	None	AMSA (2017); Lipscombe (2000)
1988	KOREAN STAR	Rocky shore, beach	Unknown, possibly reef	None	AMSA (2017); Lipscombe (2000)
1990	ARTHUR PHILLIP	Beach, mangrove	Probably reef, seagrass	Penguins: (AMSA 2017)	AMSA (2017)
1991	SANKO HARVEST	Beach	Probably reef, seagrass	Seals: Gales (1991)	AMSA (2017)
1992	ERA	Mangroves	Unknown	Mangroves: Wardrop et al. (1996) fish: Connolly & Jones (1996) sea birds: AMSA (2017)	AMSA (2017); Lipscombe (2000)
1995	IRON BARON	Beach, reef, estuary	Reef	Invertebrates, fish, algae: Edgar & Barrett (2000) penguins: Goldsworthy et al. (2000a,b); Giese et al. (2000)	AMSA (2017); Edgar & Barrett (2000); Lipscombe (2000)
1999	Port Stanvac Refinery	Beach	Unknown, possibly seagrass	None	AMSA (2000a); AMSA (2017); Lipscombe (2000)

Continued

Table 2 (Continued) Summary of habitats and flora/fauna impacted or potentially impacted by oil spills in Australia since 1970

Year	Vessel/source	Intertidal habitats potentially impacted	Subtidal habitats potentially impacted	Studies on impacts	Description of spill and/or response
2006	GLOBAL PEACE	Mangrove, seagrass, beach		Mangrove: Andersen et al. (2008) seagrass: Taylor et al. (2006); Taylor & Rasheed (2011) crabs: Andersen et al. (2008); Melville et al. (2009)	AMSA (2017)
2009	PACIFIC ADVENTURER	Beach, rocky shore		Invertebrates: Schlacher et al. (2011); Stevens et al. (2012); Finlayson et al. (2015) fish: Rissik & Esdaile (2011) wetlands: Rissik et al. (2011); Bi et al. (2011)	AMSA (2017); Schlacher et al. (2011)
2010	SHEN NIENG I	Reef, beach	Reef	Physical and antifouling impacts: GBRMPA (2011)	AMSA (2017); GBRMPA (2011)
Nearshore tropical					
1970	OCEANIC	Beach, reef, and mangroves	Unknown, possibly reef	None but extensive deaths of pearl oysters reported: Loya & Rinkevich (1980)	AMSA (2017); Manuell (1979)
1988	SIR ALEXANDER GLEN	Beach, mangrove and rocky shore	Unknown	Mangroves: Duke & Burns (1999)	AMSA (2017)
1998	Christmas Island	Beach, probably rocky shore	Unknown, probably reef	None	Dept Mines & Energy WA (1999); Lipscombe (2000)
1999	MV TORUNGEN	Beach, reef	Possibly reef	Oysters: AMSA (2017)	AMSA (2017)
2012	MV TYCOON	Beach, rocky shore, reef	Unknown, probably reef	None	AMSA (2017)
2013	FV JOSEPH M	Coral reef, beach	Coral reef	None, no impact expected	AMSA records ^a
2013	Seafaris catamaran	Coral reef, beach	Coral reef	Unknown as of November 2017	AMSA records ^a
2015	Alleged ship responsible being investigated	Coral reef, beach	Coral reef	Sea birds, turtle	AMSA records ^a , ABC (2017a)

Continued

Table 2 (Continued) Summary of habitats and flora/fauna impacted or potentially impacted by oil spills in Australia since 1970

Year	Vessel/source	Intertidal habitats potentially impacted	Subtidal habitats potentially impacted	Studies on impacts	Description of spill and/or response
Harbour/estuary temperate					
1975	LAKE ILLAWARRA	Unknown	Unknown	None	Manuell (1979)
1977	AUSTRALIS	Unknown	Unknown	None	Manuell (1979)
1977	STOLT SHEAF	Unknown	Unknown	None	Manuell (1979)
1977	YUN HAI	Unknown	Unknown	None	Manuell (1979)
1979	WORLD ENCOURAGEMENT	Mangroves	Unknown	Mangroves: Allaway (1982, 1987); Duke and Burns (1999)	AMSA (2017)
1982	Parramatta River barge	Mangrove		Mangroves: reference to crabs: Anink et al. (1985); Duke & Burns (1999)	Duke & Burns (1999)
1985	ARTHUR PHILLIP	Mangrove	Unknown	Mangroves, crustaceans and molluscs: Anink et al. (1985)	Anink et al. (1985)
1988	AL QURAIN	Harbour rock walls	None	Penguins: AMSA (2017)	AMSA (2017)
1999	LAURA D'AMATO	Beach, rocky shore	Unknown, probably rocky reef	Molluscs and other invertebrates: MacFarlane & Burchett (2003) gastropods: Reid & MacFarlane (2003) amphipods: Jones (2003)	AMSA (2000b); Lipscombe (2000); Underwood (2002)
Harbour/estuary tropical					
1987	Withnell Bay storage tank	Mangrove		Mangroves: Duke & Burns (1999)	Duke & Burns (1999)
1991	Cape Flattery storage tank	Mangrove		Mangroves: Duke & Burns (1999)	Duke & Burns (1999)
1993	Point Sampson boat harbour	Mangrove		Mangroves: Duke & Burns (1999)	Duke & Burns (1999)
1994	Cairns illegal dumping	Mangrove		Mangroves: Duke & Burns (1999)	Duke & Burns (1999)
1996	Point Sampson boat harbour	Mangrove		Mangroves: Duke & Burns (1999)	Duke & Burns (1999)
2013	Port of Brisbane	Mangrove		Cormorants, pelicans oiled: AMSA records ^a	AMSA records ^a

Continued

Table 2 (Continued) Summary of habitats and flora/fauna impacted or potentially impacted by oil spills in Australia since 1970

Year	Vessel/source	Intertidal habitats potentially impacted	Subtidal habitats potentially impacted	Studies on impacts	Description of spill and/or response
Offshore					
1975	PRINCESS ANNE MARIE	None, spill offshore	Unknown, probably none	None	AMSA (2017); Manuell (1979)
1991	KIRKI	None, spill offshore	None, spill offshore	None	AMSA (2017); Walker (1991)
1999	METAXATA	None	None	None	AMSA (2000c); Nelson (2000)
1999	SYLVAN ARROW	None, spill offshore	Unknown, probably none	None	AMSA (2017)
2001	PAX PHOENIX	None, spill offshore	Unknown, probably none	None	AMSA (2017)
2002	PACIFIC QUEST	None, spill offshore	Unknown	None	AMSA (2017)
2009	MONTARA Wellhead	Reef	Reef	Fish, birds, reptiles, mammals; PTTEP Australasia (2013) corals, seagrass: Heyward et al. (2012); Heyward et al. (2013)	Australian Government (2011); AMSA (2017); Hunter (2010)
2013	Esso COBIA offshore platform	Beach	Unknown, probably sandy seabed	None (minor spill no expected impact)	AMSA records ^a
2016	WOODSIDE COSSACK4 well	Coral reef	Unknown, probably subtidal reef or sediments	None	ABC (2017b); The West Australian (2017)
2017	Esso WEST TUNA platform	Beach	Unknown, probably sandy seabed	Unknown as of December 2017	NOPSEMA (2017); The Guardian (2017)

^a Australian Maritime Safety Authority (AMSA) records of pollution events since 2012 (involving oil spills) rated as category 3 (moderate) or above provided by the Australian Maritime Safety Authority.

PACIFIC QUEST event at the southern end of the Great Barrier Reef in eastern Australia and the 2009 MONTARA platform wellhead blowout off the north-west coast of Australia). Of these, only the MONTARA spill was subject to any detailed environmental assessment (PTTEP Australasia 2009, 2013). There have been three other spills or leaks from offshore oil and gas infrastructure: one minor spill from the ESSO COBIA platform in Bass Strait in 2013 and a recent (2017) incident from another Bass Strait platform (ESSO WEST TUNA) for which few details are available as yet; plus, the 2016 WOODSIDE COSSACK4 well leak of 10,500 litres in the Pilbara, which was not assessed for any impacts (see Tables 1 and 2).

The MONTARA H1 wellhead blowout released gas, condensate and ~4750 tonnes of crude oil into the Timor Sea 260 km offshore from 21 August to 3 November 2009. Assessments of potential impacts of this release focussed on seabirds, reptiles, commercial fish species and coral reefs (PTTEP Australasia 2013). No apparent recent mortality or disturbance impacts were detected at the nearest coral reefs (Vulcan Shoal) 27 km from the spill site. However, assessment may have begun too late to detect impacts on corals and other invertebrate fauna associated with coral reef habitats. Surveys of coral reef habitat in proximity to the MONTARA wellhead began in April 2010, some eight months after the start of the spill (Heyward 2010). Most studies examined in this review found that the most significant mortalities were immediate, if they occurred at all.

After the MONTARA spill, decreased seagrass cover was observed at Vulcan Shoal, although whether this was a natural disturbance or an effect of crude oil exposure could not be determined. In 2010, six months after the uncontrolled release, Vulcan Shoal was found to support an extensive and lush seagrass meadow (*Thalassodendron ciliatum* (32.9% cover)) (Heyward et al. 2013). However, significant declines in seagrass cover occurred in 2011 (5.2% cover), and <1% cover of seagrass was recorded on this shoal by 2013 (Heyward et al. 2013). The seagrass remnants seen in the 2011 survey (fibrous rhizomes embedded in sand and rubble areas across the central and northern plateau areas) were also further reduced by 2013 (Heyward et al. 2013). Sediment samples confirmed that Vulcan Shoal had hydrocarbons present, and this exposure, while low, was higher than other shoals examined in this study (Heyward et al. 2013). As the 2010 survey did not detect a decline in seagrass cover immediately after the spill, Heyward et al. (2013) suggested that a delayed effect from the uncontrolled release (i.e. 6–16 months post spill) seemed unlikely. Hard corals on Vulcan Shoal were normal in appearance and had not decreased in abundance during the same period, suggesting that the cause of the seagrass loss was either selective to seagrass or perhaps physical in nature (Heyward et al. 2013). There was also no evidence of accentuated algal epiphyte growth on remaining seagrasses or other nearby biota (corals) that could potentially be associated with oil spills to explain a lagged response. It was concluded that a storm or other source of strong seabed shear forces might have been responsible for the reduced seagrass cover in the 2011–2013 period (Heyward et al. 2013).

In addition to their presence in sediments at Vulcan shoals, hydrocarbons were also detected at lower levels in sediments sampled at Ashmore, Cartier and Seringapatam Reefs, the closest of which is about 120 km from the MONTARA well site. Approximately 50% of the sediment samples contained hydrocarbons five months after the spill, and ~35% of the sediment samples contained hydrocarbons 15 months after the spill (Heyward et al. 2012). Gas chromatography mass spectrometry analyses and reconstructed ion chromatogram analyses showed that the hydrocarbons had patterns typical of degraded oil, indicating that significant weathering processes had taken place. It was not possible to match the oil found in the sediments to the MONTARA reservoir (Heyward et al. 2012). The levels recorded were orders of magnitude lower than what would constitute a risk to the environment according to the ANZECC & ARM CANZ (2000) guidelines; their presence in the sediments suggests that hydrocarbons from spills can be incorporated into sediments, as can naturally released hydrocarbons. There are few sites of proven active seepage of hydrocarbons in Australia with the exception of the Timor Sea (Logan et al. 2008, Wasmund et al. 2009). However, the hydrocarbon seeps identified in the Timor Sea (Burns et al. 2010) do not appear to have a chronic or measurable effect on the quality of the marine environment (Sim et al. 2012), and consequently, it could be concluded that they do not pose a risk to animal and plant communities in the area.

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 Shoal or other surveyed locations (Heyward et al. 2010, Heyward 2011). The salient lessons from the MONTARA spill relevant to benthic macroinvertebrate and macrophyte assessment more generally include the need for baseline prespill 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. These 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 finding reinforces the need for any prespill baseline monitoring programmes to include a historical series of measurements at reference sites expected to be unimpacted in the event of a spill.

As evidenced by the PACIFIC QUEST and MONTARA spills, shipping along the Great Barrier Reef and oil and gas extraction off north-western Australia provides a potential for oil spills to impact offshore coral reefs. Corals and their larvae are sensitive to the toxic effects of oil (see later in this review). On the Great Barrier Reef, as elsewhere in the Indo-Pacific region, most coral species spawn annually over a period of just a few days following the late spring/early summer full moon (Kojis & Quinn 1982, Harrison et al. 1984, Babcock et al. 1986), and the buoyant nature of coral gametes and larvae (Harrison & Wallace 1990) places them at an elevated risk of contact with oil following spills that coincide with spawning events (and up to four weeks after). The toxicity of oil has also been demonstrated to be increased by photo-oxidation (exposure to UV light) (Negri et al. 2015), thus exacerbating the risk to coral gametes and larvae. On the Australian west coast, a larger primary mass spawning period has been reported in autumn, centred around the March full moon (Simpson et al. 1991). There is also a smaller but ecologically significant multispecific spawning period, involving fewer species and colonies, reported during late spring or early summer, often following the November full moon (Rosser & Gilmour 2008, Gilmour et al. 2009, Rosser 2013) in north-western Australia.

Subantarctic spill

Australia has experienced one Subantarctic oil spill on Macquarie Island, from the NELLA DAN in 1987, which resulted in significant impacts on marine invertebrates but seemingly minimal impacts on algae. 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 one year and seven years post impact. Pople et al. (1990) described 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 seastars) 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). Pople et al. (1990) noted a small amount of mortality of the green algae *Spongomorpha pacifica* (= *Acrosiphonia pacifica*) near the site of the heaviest oiling. Despite finding a significant difference in cover of the red alga *Palmaria georgica* between oil and control sites, they attributed the high cover of this alga at control sites to habitat differences rather than to oiling. After seven 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).

Spills in mangroves and estuaries

A moderate proportion (12/51) of oil spills in Australia have impacted mangrove habitats (Table 2); however, associated studies rarely considered impacts on fauna. After a small spill of oil from the ARTHUR PHILLIP in 1985, Anink et al. (1985) recorded up to 743 $\mu\text{g g}^{-1}$ of hydrocarbons in sediment and significant impacts on mangroves in Botany Bay. They noted that invertebrate mortality (crabs) was not evident until around four weeks post spill, and that the same observation had been made in the Parramatta River spill in 1982, with dead crabs first observed three weeks after the spill. Andersen et al. (2008) found evidence that burrowing crabs in the high intertidal area in mangroves suffered high mortality (reduced incidence of crab holes) one month following the GLOBAL PEACE spill in Gladstone harbour, but that after six months, 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 one-month post spill. Seagrass meadows were found to be not impacted by the GLOBAL PEACE spill in a study where good pre-spill spatial and seasonal baseline data existed (Taylor et al. 2006, Taylor & Rasheed 2011).

Spills on temperate and subtropical 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 invertebrates in the high intertidal zone (such as amphipods) caused from this spill, Edgar & Barrett (2000) found no evidence of oil impacts on species richness and abundance of macroalgae, reef invertebrates and fish across gradients of exposure to heavy oil which had been mixed to a sufficient depth to contact 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 one week and three months post spill, Schlacher et al. (2011) found significantly less diversity and abundance of invertebrates on impacted beaches low 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. 2011). 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, midshore 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 trochid snails (*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 eight years prior to the spill. During 2000, MacFarlane & Burchett (2003) documented some recovery at all sites 12 months post spill. They did not note any significant effects on macroalgae from the spill. A follow-up laboratory study (Reid & MacFarlane 2003) confirmed the toxicity of oil similar to that released from the LAURA D'AMATO on *Austrocochlea*. The same oil spill caused near total mortality of the amphipod *Exoedicerus* sp. on sandy beaches affected by the spill (Jones 2003). Rates of recovery tended to be most rapid at the less impacted site, which recovered after four months compared to more heavily impacted sites, some of which showed no recovery after nine 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 three to five months.

Oil spill simulation field studies in Australia

Few studies simulating the effects of an oil spill in natural habitats have been undertaken in Australia. Clarke & 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 of mortality among high intertidal gastropods (*Littorina*, *Bembicium*, *Salinator* and *Ophicardelus*) and crabs. Similar effects were found 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 salt marsh 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 (one part oil agitated with two parts seawater) Dubai light crude oil. High mortality of the gastropods *Assiminea*, *Cassidula* (= *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 and the effect of dispersants but did not examine the influence on mangrove invertebrates (Burns et al. 1999, 2000). Similarly, in the temperate Gulf St Vincent in South Australia, Wardrop et al. (1987) measured the rates of mangrove defoliation and recovery after application of Arabian light crude and Tirrawarra crude oils with and without dispersant. They found increased levels of initial defoliation on mangroves in treatments with dispersants but no long-term negative impacts, although they did not examine any effects on biota associated with mangroves.

Thompson et al. (2007) conducted a study in Antarctica using synthetic lubricants to assess the effect of recolonisation in defaunated plots in marine sediments. They found that abundances of recently colonised animals in plots were the same five weeks post oiling, but 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).

*Measured responses of Australian invertebrates, seagrasses
and macroalgae to oil toxicity in laboratory studies*

Marine invertebrate toxicity responses

Experimental assays of oil and dispersant toxicities in Australia are limited to a few studies. Gulec et al. (1997) measured the 96-hour 50% lethal concentrations (96-hour LC50 s) for the amphipod *Allorchestes compressa* of 311,000 ppm (WAF of Bass Strait crude oil diluted in seawater), 16.2 ppm (oil dispersed 9:1 with Corexit 9500 and diluted in seawater) and 14.8 ppm (as previously for Corexit 9527) indicating that the oil alone (without dispersant) was much less toxic. It should be noted that these are not the concentrations of hydrocarbons but the concentrations of WAF. In the case of Gulec et al. (1997), the WAF was obtained by stirring one part crude oil and nine parts water for 24 hours and then, after settling for one hour, siphoning off the WAF from below the oil. In the previous example, 311,000 ppm is 311 mL WAF and 689 mL seawater in each litre. They also demonstrated 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.

Despite the importance of coral reefs in Australia, there have been few studies of the response of corals and coral larvae to oil in Australia. Branches of *Acropora muricata* (= *Acropora formosa*) exposed to marine fuel oil showed complete living tissue disintegration after 48 hours (Harrison et al. 1990). Harrison et al. (1990) further reported that the *A. muricata* colonies expelled zooxanthellae and showed enhanced production of mucus within the first hour of exposure to marine fuel oil, mortality after 12 hours exposure, and an increase in the concentration of pigmented bacteria on the mucus after 48 hours.

Most broadcast spawning corals have buoyant eggs and larvae (Harrison & Wallace 1990), making them potentially highly vulnerable to oil spills (Harrison 1999). The WAF of heavy crude oil from the Wandoo reservoir off north-western Australia did not inhibit fertilisation of *Acropora millepora* gametes. However, fertilisation was disrupted by the dispersant Corexit 9527 and the dispersed oil fraction (Negri & Heyward 2000). Negri & Heyward (2000) further showed that larval metamorphosis was inhibited by exposure to crude oil, the dispersant Corexit 9527 and the dispersed oil fraction. Moreover, although the crude oil and dispersant inhibited larval metamorphosis individually, this toxicity was magnified when the larvae were exposed to combinations of both (i.e. dispersed oil fraction). A similar result was reported in three different species (*Acropora tenuis*, *Platygyra sinensis*, *Coelastrea aspera* [= *Goniastrea aspera*]); significantly increased larval mortality rates in the dispersant (Ardrox 6120) and dispersed bunker fuel oil 467TM treatments were recorded compared to seawater controls (Lane & Harrison 2000). Increased exposure time to oil resulted in increased toxic effects, and the dispersed oil was the most toxic of the contaminants tested due to an increase in bioavailability of hydrocarbons as a result of the dispersal (Lane & Harrison 2000). Of the three species, significant mortality was only measured in *C. aspera* larvae when exposed to treatments of different concentrations of the oil WAF alone. However, the experiment was hampered by low rates of survivorship in the controls.

Natural gas condensate can be considered as a light crude oil, and the WAF of Browse Basin condensate (north-western Australia) was shown to inhibit metamorphosis of coral (*Acropora tenuis*) and sponge (*Rhopaloeides odorabile*) larvae (Negri et al. 2015). Coral larvae were 100 times more sensitive (inhibition at 100 µg L⁻¹ polycyclic aromatic hydrocarbons [PAHs]) than sponge larvae (10,000 µg L⁻¹). When the WAF was exposed to ultraviolet (UV) light, the sensitivity of coral larvae, but not sponge larvae, increased by 40% (Negri et al. 2015). Oil exposed to ultraviolet radiation in surface waters may photo-oxidise (Pelletier et al. 1997), which may substantially increase the toxicity and bioavailability of hydrocarbon components such as PAHs (Ehrhardt & Burns 1993, Dutta & Harayama 2000, Neff 2002).

Unfortunately, studies of experimental laboratory and field-based oil toxicity are not directly comparable given the use of different field/laboratory conditions, weathering techniques (with/without predistilling, and WAF varying between two and nine parts seawater), LC50 calculations (ppm total hydrocarbons versus % WAF), and time periods (24–96 hours). The use of regression analyses to predict LC50 s from experimental data also makes comparison difficult, and applications to field situations even more challenging. As described previously, 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 LC50 s 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 seastars and a gastropod. Ryder et al. (2004) found that the herbivorous seastar *Parvulastra exigua* (= *Patiriella exigua*) from Port Phillip Bay avoided oiled sediment in the laboratory and in doing so was able to avoid its

narcotising effects. The ability of the predatory Port Phillip Bay seastar *Coscinasterias muricata* to locate prey was significantly reduced when exposed to WAFs of Bass Strait crude oil with and without added dispersant; however, seastars 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 *Conuber conicum* (= *Polinices conicus*) after 30 minutes of exposure to dilutions of Bass Strait crude oil WAF, 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).

Seagrass toxicity responses

Australia has some of the most extensive seagrass meadows in the world, which are of great importance to coastal stability and as a food source and nursery for coastal invertebrates, fishes, mammals and reptiles (Larkum et al. 1989, Butler & Jernakoff 1999). Despite the importance of these habitats, there have been few studies on the toxic response to oil of seagrasses in Australia. The first such study was carried out by Hatcher & Larkum (1982), who found exposure of *Posidonia australis* to 6800 ppm of Bass Strait crude oil and 120 ppm of the dispersant Corexit 8667 resulted in decreased photosynthesis and increased respiration, but that leaf production was not affected. The dispersed oil mixture induced a greater physiological stress than the oil on its own. However, Ralph & Burchett (1998) found photosynthesis and pigments in *Halophila ovalis* showed only a minor negative response when exposed to 1% weight/volume (w/v) Bass Strait crude oil and an oil: Corexit 9527 mix (10:1).

More recently, Macinnis-Ng & Ralph (2003) showed Champion crude oil (0.25% w/v) had an effect on photosynthesis and pigments of *Zostera capricorni* in the laboratory; however, *in situ* field trials in enclosures demonstrated reduced impacts. It was concluded that laboratory trials may overemphasise toxic effects. In another study on *Z. capricorni*, Wilson & Ralph (2012) showed no impact at concentrations less than 0.4% of the water-soluble fraction (WSF) of Tapis crude oil. Thus, while such studies highlight the potential for toxicity following exposure to oil and oil and dispersant mixes, consensus on impacts to the overall health of seagrass is lacking due to a high degree of variability in oiling scenarios and potentially response among seagrass species. This remains an issue for both the Australian and international context (Fonseca et al. 2017).

Macroalgae toxicity responses

Measurements of toxicity of Australian macroalgae to oil and dispersants are limited to the measurements made by Burrige & Shir (1995). In these experiments, they examined the effects on germination of the brown alga *Phyllospora comosa* and found that the 48-hour EC50 (maximal effective concentration that induces a response halfway between the baseline and maximum germination inhibition after 48 hours) was 130 ppm for Bass Strait crude oil and 6800 ppm for diesel fuel.

Application of two dispersants (Corexit 7664, 8667A) resulted in substantially decreased toxicity of the crude oil to a 48-hour EC50 of 2500 and 4000 ppm, respectively, while little change in toxicity was evident with use of two other dispersants (Corexit 9500 and 9527). All four dispersants increased toxicity of diesel (48-hour EC50 of 340–420 ppm) (Burrige & Shir 1995).

Impacts of oil on marine invertebrates, seagrass and macroalgae

Acute and toxic impacts of oil on marine invertebrates

Impacts on zooplankton and invertebrate 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 an oil spill on a population or a 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 an oil spill event. For example, Beiras & 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 4-fold dilution this WAF was toxic to the larvae immediately after the oil spill, and toxicity to larvae from the undiluted contaminated seawater persisted in seawater collected two months after the spill.

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 in the United States, which killed numerous adult marine invertebrates, and noted that the diesel spill had occurred during the spawning season for many species. He then tested the larvae of 14 species of echinoderms (seastars and a sea urchin), gastropods, a bivalve and a chiton mollusc, annelid worms, a barnacle and an ascidian in a 0.5% oil–water mixture. He found that while all larvae in control conditions survived, all larvae in the oil–water mixture died within three hours to three days, other than those of one seastar *Crossaster*, which all had died after eight days (Chia 1973). Unlike the other invertebrates, *Crossaster* has large yolky eggs with lecithotrophic (non-feeding) development, which may be a reproductive strategy that is resilient to oiling. Around the same time, Wells (1972) also demonstrated the toxic effect of oil on lobster (Crustacea) larvae (Wells 1972, Wells & Sprague 1976, Stejskal 2000). Byrne & 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 cause mortality of sponge recruits (Cebrian & Uriz 2007). For zooplankton, Elmgren et al. (1983) found ostracods and harpacticoid copepods were significantly impacted by the 1977 TSEIS spill in Sweden. 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 31.4 ppm WAF. In other studies, Almeda et al. (2014a,b) demonstrated that dispersed oil is more highly toxic than crude oil alone.

Following the Gulf of Mexico DEEPWATER HORIZON spill in 2010, settlement of the commercially important crab *Callinectes sapidus* along the Mississippi coast was assessed (Fulford et al. 2014). Natural settlement rates of this species vary considerably each year, and they did not detect any change attributable to the oil spill. They also noted that concentrations of oil known to be toxic to crab larvae were not experienced in locations important for crab settlement, at least in their study area.

Studies consistently indicate that the impact of oil on larvae is largely dependent on exposure concentration (Almeda et al. 2013, Fulford et al. 2014). Thus, the risk of exposure to toxic concentrations is a critical element when evaluating likelihood of impact. This emphasises the importance of studies that determine at what concentrations hydrocarbons cause mortality or have significant sublethal impacts on 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 context becomes very important. This underlines the importance of studies that not only provide data in a context relevant to Australia, but does 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 large unconfined spills, where an advancing plume may create an exposure risk to animals in intertidal and subtidal habitats.

Impacts on adult invertebrates

Early reports on the impacts of oil spills generally relate to intertidal exposures following shipping or refinery accidents. These indicated that a very wide range of marine invertebrate taxa are affected by oil, with very high potential mortalities (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 provide examples below of the available information on how different groups of marine invertebrates are affected by oil exposure. Although the following analysis suggests a range of responses between taxa, it is likely the severity of a spill and the levels of exposure (not the type of animal) are probably more important in determining the level of mortality that occurs.

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 where extensive oil and gas exploration and production is occurring. There are few reports of sponge mortality from oil spills in the international literature, and those that do exist make it unclear whether sponges are highly vulnerable to oil toxicity or not. 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 by 2011, as shown by annual monitoring to 2011 (Castège et al. 2014). However, it is possible that the high-pressure hot-water cleaning of the site contributed to the mortality and/or the absence of recovery in this instance (Castège et al. 2014). Rocky shore monitoring in Milford Haven, Wales, six months before and after the 1996 SEA EMPRESS spill did not detect any effects on abundance of *Hymeniacidon perlevis* in areas affected by the spill but not cleaned, suggesting this sponge at least is not especially vulnerable to the toxic effects of oil (Moore 1997). Batista et al. (2013) determined that *Hymeniacidon heliophila* was a good indicator of 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 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 North West Shelf (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 refinery oil spill in Panama. In the absence of any other studies, an understanding of the response of Australian bryozoa to oil is needed.

Cnidarians As a group, anthozoan scleractinian 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, Guzmán & Holst 1993, Guzmán et al. 1994), as well as chronic sublethal effects (see reviews by Johnson 1977, Loya & Rinkevich 1980, Fucik et al. 1984, Suchanek 1993). Direct coating of corals by oil have been reported to result in stress and mortality of coral colonies (Loya & Rinkevich 1979, 1980, Yender & Michel 2010). Rinkevich & Loya (1977) reported that a greater number of *Stylophora pistillata* colonies died on hydrocarbon-affected reefs after one year (~42% mortality) compared to control reefs (~10% mortality) in Eilat, Israel. Coral mortality was also recorded following a major oil spill in Panama (70% Venezuelan, 30% Mexican Isthmus crude oil), and the total coral cover in heavily oiled reefs declined by between ~55% and ~75% at 3–6 m and <3 m depths, respectively (Jackson et al. 1989). A significant decline in coral cover was recorded at ≤3 m depth, and most corals at >3 m depth that remained alive after the spill showed signs of stress that included tissue swelling, conspicuous production of mucus and recently dead areas devoid of live coral tissue (Jackson et al. 1989).

Laboratory studies have also demonstrated stress and mortality of coral colonies exposed to oil. Johannes et al. (1972) exposed 22 species of coral at Eniwetok Atoll to Santa Maria crude oil for 1.5 hours and showed a complete breakdown of tissue on the areas to which crude oil adhered, whereas no visible effects were evident in the areas to which the oil did not adhere to the coral colonies. Reimer (1975) exposed colonies of *Pocillopora* cf. *damicornis* to Bunker-C and marine diesel oils and reported that all colonies lost almost all living tissue after 16 days, while the control colonies sustained >95% living tissue. Colonies of other genera (*Psammocora* and *Porites*) were more resilient, surviving for longer, but died after 114 days. However, this was done under extreme conditions with corals in some cases being fully submersed in oil. Similarly, branches of *Acropora muricata* (= *Acropora formosa*) exposed to marine fuel oil showed complete living tissue disintegration after 48 hours (Harrison et al. 1990). Harrison et al. (1990) further reported that the *A. muricata* colonies expelled zooxanthellae and showed enhanced production of mucus after 12–24 hours. While no significant impact on the survivorship from the WAF of Egyptian crude oil was reported in *Stylophora pistillata* and *Pocillopora damicornis* following exposure, significant mortality was reported following exposure to dispersants (Emulgal C-100, Dispolen 36S, Inipol IP-90, Petrotech PTI-25, Slickgone NS, Bioreico R-93) and the dispersed oil fraction (Shafir et al. 2007).

Laboratory and field studies have demonstrated that branching corals appear to have a higher susceptibility to hydrocarbon exposure than massive corals or corals with large polyps. After exposure to Santa Maria crude oil, branching *Acropora* and branching *Pocillopora* showed the highest susceptibility to coating and retention, and oil was still visible on the corals after four weeks (Johannes et al. 1972). Corals with large fleshy polyps and abundant mucus (*Fungia* and *Lobophyllia* [= *Symphyllia*]) retained almost no oil and showed no damage, and *Astreopora*, *Favia*, *Favites*, *Montipora*, *Plesiastrea*, *Porites*, *Psammocora* and *Turbinaria* showed intermediate effects (Johannes et al. 1972). The increased susceptibility of branching coral species was further reinforced by Guzmán et al. (1991), whereby following a major oil spill in Panama (70% Venezuelan, 30% Mexican Isthmus crude oil), the branching coral *Acropora palmata* suffered far greater mortality compared to the massive species that were common on the local reefs.

Based on the literature, those genera capable of producing large amounts of mucus may be less susceptible to damage from exposure to oil. Production of mucus is generally a stress response (Beeden et al. 2008, Erfteimeijer et al. 2012) and may aid removal of oil and toxicants and provide these corals with a level of resilience. However, excess production of mucus resulting from hydrocarbon exposure may also lead to enhanced bacterial growth and degradation of the coral tissue (Loya & Rinkevich 1980, Peters et al. 1981) as well as being energetically costly (see Davies & Hawkins 1998 for review).

The reported responses of other cnidarians to oil vary considerably in the literature. 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 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 sublethal effects occurred at lower concentrations. 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 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 persisting five years after the event (Burns et al. 1993). However, Castège 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 TAMPICO MARU spill in Mexico that killed most 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 potential 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 (*Minuca pugnax* [= *Uca pugnax*]) suffered high mortality. Stomatopods were heavily impacted in intertidal seagrass beds after the 1986 refinery spill at Galeta, Panama (Jackson et al. 1989). Following the 1977 TSEIS 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). Amphipods, isopods and crabs were also heavily impacted by the 1978 AMOCO CADIZ spill in France (Chassé 1978, O'Sullivan 1978, Conan 1982), with populations of numerous amphipod species remaining at low levels or absent 10 years later (Dauvin 1987, Dauvin & Gentil 1990).

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 verrucosa* [= *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; Castège et al. 2014). Conversely, after the West Falmouth spill, the fiddler crab population had still not recovered seven years post spill because the habitat was still contaminated (Krebs & Burns 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, Chassé (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 *Semibalanus cariosus* (= *Balanus cariosus*) 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 et al. 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 five years (Burns et al. 1993). Castège et al. (2014) reported a similar time (two to five years) for the tunicate *Botryllus schlosseri* to reappear at their study site in France after the 2002 PRESTIGE oil spill. Few reports exist on the impacts of oil on tunicates. Nevertheless, given their importance as filter feeders and the abundance of some species among intertidal and benthic assemblages in eastern Australia (e.g. *Pyura stolonifera*, Dakin 1960), and among subtidal habitats on the Australian North West Shelf (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 65 spill in Massachusetts, USA, where a large number of *Alitta virens* (= *Nereis virens*) were killed among numerous other marine invertebrates (Hampson & Moul 1978). Elmgren et al. (1983) reported polychaetes (*Byligides sarsi* [= *Harmothoe sarsi*]) were reduced by 95% in sediments contaminated by the 1977 TSEIS oil spill in the Baltic Sea and that turbellarians and kinorhynchans were also significantly affected, but 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 with differing responses amongst individual species. The reason that some species (e.g. *Daptonema trabeculosum*) were very sensitive to the oil, while others (e.g. *Spirinia gerlachi*) were resilient could not be established (Beyrem et al. 2010). Nematodes in deep-water (~1200 m) samples were found to respond positively to the 2010 DEEPWATER HORIZON spill in the Gulf of Mexico (Montagna et al. 2013). It was hypothesised that the nematodes may have responded to enhancement of the bacterial flora through oil-induced organic sediment enrichment and reduction of competitive species in taxa negatively impacted by the spill, such as copepods. The ratio of nematodes to copepods has previously been proposed as an indication of pollution impact (Raffaelli & Mason 1981), although the effectiveness of its use has been subject to extensive debate (e.g. see Amjad & Gray 1983, Raffaelli 1987). Dexter (1984) reported reductions in polychaetes after an oil spill in Australia, but probable impacts of oil spills on worms in the Australian situation are not generally known.

Echinoderms Echinoderms are among the most vulnerable of marine invertebrates to oil spills, and many early studies documenting 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). Castège 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 two to four years to recover. On the other hand, the seastar *Asterina gibbosa* and a holothurian (*Holothuria* sp.) were either minimally or not impacted by the spill. Conan (1982) reported severe mortality of 1 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 *Lytechinus variegatus* were killed in the experimental treatment areas. 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 (Johnson 1977).

Australia's marine ecosystems harbour a high level of echinoderm diversity, including in areas where petroleum exploration and extraction, as well as a high level of commercial shipping, also occur. For example, one live-bearing intertidal species of seastar in the Great Australian Bight (*Parvulastra parvivipara*) has a species distribution of <200 km (Edgar 2012), a very small range for a marine species. A seastar with similar brooding habit, *Asterina phylactica*, suffered significant mortality following the 1996 SEA EMPRESS spill in Wales (Moore 2006). On the Australian North West Shelf, 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 for any country. Two studies have been conducted on sublethal impacts of oil on seastars 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 due to oil spills, either due to the oil itself or inappropriate use of dispersants (e.g. Smith 1968, Mitchell et al. 1970, Chia 1971, Woodin et al. 1972, Southward & Southward 1978, Le Hir & Hily 2002). Mortality rates are dependent on degree of exposure, which in turn is often associated with shore height in intertidal populations. Following the 1978 AMOCO CADIZ spill in France, Chassé (1978) and O’Sullivan (1978) documented high mortality of gastropods *Littorina*, *Gibbula* and *Monodonta* and to a lesser extent limpets of the genus *Patella*. 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 heavy intertidal oiling has occurred, mussels 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). Contrary to the situation with the AMOCO CADIZ spill, very heavy mortality of *Patella* followed the 1967 TORREY CANYON spill in Cornwall in the United Kingdom; however, most of this was attributed the toxicity of the first-generation dispersants used (Smith 1968, Southward & Southward 1978, Hawkins & Southward 1992, Hawkins et al. 2017b).

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 nuttallii* and *Macoma* spp.) and gastropods (*Acmaea* spp.), while oysters (*Magallana gigas* [= *Crassostrea gigas*]), mussels (*Mytilus edulis*) and the gastropods (*Littorina scutulata* and *Littorina sitkana*) experienced little or no mortality. Predatory whelks (*Thais* spp.) were found with moribund appearance but recovered when returned into clean seawater. In another study on the same spill, Chia (1971) found extensive mortality of marine invertebrates including limpets and chitons but noted that two species of littorinid periwinkles 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 more than two years post spill, 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 eight to nine 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, the gastropod *Cerithium* has been reported to continue to feed on oiled intertidal flats (Spooner 1970, Chan 1977). After the 1977 TSEIS oil spill in the Baltic Sea, Elmgren et al. (1983) showed that despite oil contamination of sediment killing 95% of amphipods and polychaetes, the clam *Limecola balthica* (= *Macoma balthica*) experienced minimal mortality despite becoming highly contaminated (to a level of 2 mg g⁻¹ 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 mussels and oysters. Bivalve molluscs in particular are effective at bioaccumulation of these toxicants (La Peyre et al. 2014), and so can potentially suffer from 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.

Acute and toxic impacts of oil on marine macrophytes

Seagrasses and macroalgae are considered vulnerable to oil spills because they are located in nearshore subtidal and intertidal habitats, which are the areas where uncontained oil tends to accumulate following its release. Over 75% of oil spills in the United States are estimated to occur within coastal waters, specifically estuaries, enclosed bays and wetlands (Kennish 1992). Despite this risk, and the demonstrated toxicity of oils to marine plants (Hatcher & Larkum 1982, Thorhaug et al. 1986, Ralph & Burchett 1998, Wilson & Ralph 2012, Lewis & Pryor 2013), cases of large-scale losses of seagrasses and macroalgae following an oil spill are restricted to a few isolated incidents (Nelson-Smith 1973, Floc'h & Diouris 1980, Jackson et al. 1989). This, in part, appears to be a reflection of environment, with geomorphologic and nearshore hydrodynamic conditions likely to influence the behaviour of oils such that the effects in a sheltered bay with low energy and little water exchange will be different to the effects on an exposed rocky coast with high energy and strong current (Taylor & Rasheed 2011). Understanding such considerations is important, as responses to oil spills need to be tailored to suit conditions at the local scale.

As most of the constituents of petroleum oil have low solubility in water, buoyant surface plumes are generally transported directly to intertidal habitats, often bypassing subtidal habitats and reducing the levels of exposure. Accordingly, marine flora that occur in the shallow intertidal zone are more susceptible to large-scale oil spills than flora that occur subtidally (Lewis & Pryor 2013). Following the 1986 Galeta oil spill in Panama (>8 million litres of crude oil spilled into a complex region of mangroves, seagrasses and coral reefs just east of the Caribbean entrance to the Panama Canal), Jackson et al. (1989) reported loss of entire beds of the seagrass *Thalassia testudinum* (a genus also common in tropical areas of Australia) on heavily oiled reef flats, while subtidal meadows in the same area survived. Substantial macroalgal losses following oiling of intertidal habitats were also reported by Floc'h & Diouris (1980) and Nelson-Smith (1973), while more recently there have been anecdotal accounts of large-scale losses of macroalgae in intertidal seaweed farms in Timor following the MONTARA oil spill (Mason 2011). It is important to note, however, that some of the decline reported by Floc'h & Diouris (1980) was attributed elsewhere to indirect effects associated with grazing of herbivores, rather than from direct hydrocarbon toxicity associated with crude oil impacts on macroalgae (Laubier 1980).

Although in many reported cases subtidal seagrass or macroalgal assemblages have not been severely affected following oil spills (Kenworthy et al. 1993, Dean et al. 1998, Edgar et al. 2003, Taylor et al. 2006, Taylor & Rasheed 2011), subtidal assemblages can be subjected to, and impacted by, direct contact with oil under certain environmental conditions. For example, subtidal beds of the seagrass *Thalassia testudinum* were decimated following a crude oil spill in Puerto Rico in 1973, where strong weather conditions caused the entrainment of oil into the seagrass meadow (Nadeau & Bergquist 1977). In this instance, the ship's master intentionally released 37,579 barrels (5.98 million litres) of Venezuelan (Tijuana) crude oil (following a vessel grounding), of which an estimated 24,000 barrels (3.82 million litres) was stranded within nearby coastal ecosystems (Hoff & Michel 2014). While the crude oil was considered to be of low toxicity, the impacts to the seagrass were so severe that even the rhizome layer was affected (Nadeau & Bergquist 1977).

Despite the potential for more severe consequences, sublethal responses (e.g. localised necrosis) have more commonly been reported following exposure to seagrasses and macroalgae in subtidal depths (Jackson et al. 1989, Taylor & Rasheed 2011). This suggests that environmental factors can act to reduce the potential for toxicity of oil in subtidal environments. For example, Taylor & Rasheed (2011) reported that following a spill of ~25 tonnes (~28,000 litres) of bunker oil in the subtropical Port of Gladstone, Queensland, in 2006, no significant differences were detected in seagrass shoot biomass or seagrass extent (i.e. area affected) between exposed and 'clean' seagrass meadows. This lack of impact was suggested to result from several extenuating factors: the oil spill occurred on a high neap tide; consequently, the intertidal meadows were probably not exposed until

two to three days post spill, by which time the oil would have spread and thinned, with evaporation, photooxidation and dissolution processes likely reducing the quantity of remaining oil in the area (Taylor & Rasheed 2011) and reducing its acute toxicity.

Effects of direct contact of oil on seagrasses and macroalgae

Smothering, fouling and asphyxiation are some of the physical effects that have been documented from oil contamination in marine plants (Blumer 1971, Cintron et al. 1981). In macroalgae, oil can act as a physical barrier for the diffusion of CO₂ across cell walls (O'Brien & Dixon 1976), although the mucilaginous slime covering many large brown algae is considered a protective device against coating by oil. For example, following the 1969 PLATFORM A blowout off Santa Barbara, California, oil retained in the canopy of giant kelp *Macrocystis* beds adhered tightly to blades reaching the surface, but removal of the oil layer revealed healthy tissue beneath (Mitchell et al. 1970). Observable damage to the macroalgae was negligible, likely because of secretion of mucus (Mitchell et al. 1970). A lack of damage to *Macrocystis* was also reported following exposure to Arabian crude oil spilled from the tanker METULA, wrecked in the Magellan Straits (Wardley-Smith 1974). In addition to gas exchange problems, macroalgae, and likely some small annual seagrass species, can become seriously overweighted and subject to breakage by waves (Nelson-Smith 1973) or other hydrodynamic processes following exposure to oil. A strong oil adsorptive capacity has been noted for many macroalgal species (including *Ascophyllum nodosum*, *Fucus* spp., *Pelvetia canaliculata*, *Hesperophycus californicus* [= *Hesperophycus harveyanus*], *Mastocarpus stellatus* [= *Gigartina stellata*] and *Gelidium crinale*; O'Brien & Dixon 1976), and the seagrass *Phyllospadix torreyi* (Foster et al. 1971), and this is likely to be exacerbated where the oil strands on a rising tiding contacting partially desiccated plants.

Toxicity effects on seagrasses

When seagrass leaves are exposed to petroleum oil, sublethal quantities of the WAF can be incorporated into the tissue, causing a reduction in tolerance to other stress factors (Zieman et al. 1984). The toxic components of oil are thought to be the PAHs, which are lipophilic and tend to accumulate in the thylakoid membranes of chloroplasts (Ren et al. 1994). Consequently, thylakoid membrane oxidation impacting on photosynthesis is a symptom of oil toxicity (Ren et al. 1994, Marwood et al. 1999). The type of oil spilled has different effects on different aquatic plants, and the use of dispersants can also contribute to overall impact (Thorhaug 1988).

Direct contact with the above-ground biomass of seagrass following oil exposure can result in morphological changes (Jacobs 1988). Seagrass blades can become bleached, blackened, yellowed or detached from the plant following direct oil contamination (den Hartog & Jacobs 1980, Jackson et al. 1989, Dean et al. 1998), while other effects from direct contact include a decrease in the density of vegetative and flowering shoots (den Hartog & Jacobs 1980, Jackson et al. 1989, Dean et al. 1998). There are, however, many reported cases where hydrocarbon exposure has not led to any form of physical change in seagrasses (Kenworthy et al. 1993, Dean et al. 1998, Taylor et al. 2006, Taylor & Rasheed 2011). For example, approximately one year after exposure during the Gulf War spill, Kenworthy et al. (1993) found no measurable impact on species composition, distribution, abundance, net production or growth of the annual seagrass species *Halodule uninervis*, *Halophila ovalis* or *Halophila stipulacea*.

Studies of exposure of oil and dispersants to seagrasses have revealed a range of responses with some species more tolerant than others. *Thalassia testudinum* has been used in a number of studies from the Caribbean (Baca & Getter 1984, Thorhaug et al. 1986, Ballou et al. 1987, Thorhaug & Marcus 1987a,b) and is generally less sensitive to dispersed oil exposure (LD50—lethal dose causing 50% mortality of 1.25 mL L⁻¹) than other seagrasses such as *Halodule wrightii* and *Syringodium filiforme* (LD50 of 0.75 mL L⁻¹). In other studies, seagrasses have been shown to be less sensitive to unmixed oil exposure than to dispersed oil. In support of the field observations of Kenworthy et al.

(1993) discussed previously, Durako et al. (1993) found that a 1% weight/volume mix of crude oil in seawater had no detectable effect on *Halodule uninervis*, *Halophila stipulacea* and *Halophila ovalis*. In Australia, the same concentrations of oil and an oil/dispersant mix also had negligible impacts on *H. ovalis* (Ralph & Burchett 1998). From these studies, it appears that a WSF of between 0.25%–1% (w/v) solution of crude oil in seawater is sufficient to induce physiological responses; however, valid generalisations concerning the magnitude of phytotoxicity for crude and refined oils are difficult to make due to the uneven nature and wide range of reported toxic effect concentrations (Hatcher & Larkum 1982, Thorhaug et al. 1986, Thorhaug 1988, Ralph & Burchett 1998, Lewis & Pryor 2013). Further, some species are more sensitive to oil exposure than others (Thorhaug et al. 1986), although the reasons for this remain unclear. It is, therefore, difficult to compare results from studies testing different oils on different species; any one of these variables, or a complex interaction between them, could be responsible for the variable results (Runcie et al. 2005).

Phytotoxic effect of petroleum oil on seagrasses can apparently lead to a range of sublethal responses including reduced growth rates (Howard & Edgar 1994), bleaching, decrease in the density of shoots, reduced flowering success (den Hartog & Jacobs 1980, Dean et al. 1998) and blackened leaves that may detach from the plant (den Hartog & Jacobs 1980). Direct exposure, however, does not always induce toxic effects (Kenworthy et al. 1993, Dean et al. 1998), even under laboratory conditions (Wilson & Ralph 2012). The disparity among research findings may be due to different experimental methods, including the range of indicators used, varying exposure and temperature regimes, and the specific petrochemicals evaluated (Lewis & Pryor 2013). Further, morphological variation between seagrass species is considerable, and species resilience to petrochemical impacts is likely to reflect morphological traits.

Published studies of exposure to oils exist for nine of the world's 72 seagrass species (Lewis & Pryor 2013), with congeners of all nine species present in Australian waters. Thorhaug et al. (1986) showed clear differences in the response of different species of tropical seagrass to oil, but other research to date has largely been conducted on single species (Hatcher & Larkum 1982, Ralph & Burchett 1998, Macinnis-Ng & Ralph 2003). While it is difficult to generalise for these reasons, the seagrass *Thalassia testudinum* has been consistently more tolerant to crude oils and dispersed oils than other species that occur in tropical waters, such as *Halodule wrightii* and *Syringodium filiforme* (Thorhaug et al. 1986). *Halophila* spp. have also shown a degree of tolerance to petrochemical exposure (up to 1% (w/v) solution of Bass Strait crude oil) under laboratory conditions (Ralph & Burchett 1998) and during field testing (Kenworthy et al. 1993).

Stress conditions associated with oil exposure to seagrasses can reduce the rate of photosynthesis, block photosynthetic electron transport or disturb the pigment-protein apparatus (Maxwell & Johnson 2000). When light energy is absorbed by a plant, some is used for photochemical reactions, but a proportion is emitted as heat or fluorescence. If the functional state of the photosynthetic apparatus changes, the amount of fluorescence emitted also changes. This information can be used to quantify a stressor (Maxwell & Johnson 2000). In this way, chlorophyll-a fluorescence can provide important physiological data on the effect of hydrocarbon exposure on photosynthetic activity (Ralph & Burchett 1998, Macinnis-Ng & Ralph 2003, Wilson & Ralph 2012).

Photosynthetic stress of seagrasses is typically monitored using chlorophyll-a fluorescence (specifically the effective quantum yield of Photosystem II ($\Delta F/F_m'$) using pulse amplitude modulation techniques), but also via analyses of chlorophyll-a pigment concentrations. The value of $\Delta F/F_m'$ provides information regarding photosynthetic activity and thus physiological health of the seagrass. In laboratory experiments, Wilson & Ralph (2012) reported minimal, if any, long-term change to $\Delta F/F_m'$ in the seagrass *Zostera capricorni* when exposed to crude oil but found short-term increases similar to stimulatory effects observed in other oil exposure studies (Karydis & Fogg 1980, Chan & Chiu 1985). Oil taken up by the leaf blades was suggested to lead to a short-term stimulatory effect (hours) before returning to typical rates of photosynthesis (Wilson & Ralph 2012). In other studies, toxic impacts (declines in photosynthesis) of oils were detected using $\Delta F/F_m'$ in seagrass

(Ralph & Burchett 1998, Macinnis-Ng & Ralph 2003), freshwater macrophytes (Marwood et al. 2001), phytoplankton (Marwood et al. 1999) and corals (Jones & Heyward 2003). For seagrasses, Marwood et al. (2001) concluded that oil can cause the inactivation of Photosystem II reaction centres due to oxidation or degradation of D1 proteins.

Toxicity effects on macroalgae

It has been known for at least 60 years that crude and refined oils are phytotoxic to algae (Currier & Peoples 1954, Van Overbeek & Blondeau 1954; cited in Lewis & Pryor 2013), with water-soluble hydrocarbon molecules more toxic to macroalgae than larger molecules (Van Overbeek & Blondeau 1954, Kauss et al. 1973, cited in O'Brien & Dixon, 1976). There are several reported cases of toxicity associated with oil inducing morphological change in macroalgae (Lewis & Pryor 2013). The sequential effects of an oil spill on intertidal algae have been recorded in a heavily oiled cove near to the wreck of the AMOCO CADIZ in Brittany, France, in 1978 (Floc'h & Diouris 1980). This release involved ~1.6 million barrels (~250,000 tonnes) of light crude oil produced from Ras Tanura, Saudi Arabia. Fresh oil initially acted like a contact herbicide, causing intensive bleaching of the most sensitive thalli of green algae (Floc'h & Diouris 1980). This was followed by localised necrosis in red algae and widespread necrosis in brown algae (Floc'h & Diouris 1980). More recently, Stekoll & Deysher (2000) reported physical injury to the macroalga *Fucus distichus* (= *Fucus gardneri*) following the 1989 EXXON VALDEZ oil spill. This resulted in lower biomass, lower percent cover, impaired reproductive capability and alterations to population structure (as evidenced by differences in densities and proportions amongst size classes). However, in this case, injuries may also have been caused by the oil spill clean-up (Stekoll & Deysher 2000).

Oil concentrations with observed toxic effects on algae vary greatly among species and studies, ranging from 0.002 to 10,000 ppm (Lewis & Pryor 2013). Indeed, algae show a variable response to the effects of oil under the same conditions in comparative studies. For example, of 10 species of Norwegian algae (four Phaeophyta, three Rhodophyta and three Chlorophyta) monitored for two years under mesocosm conditions with continuous exposure to 0.129 ppm of diesel (129 µg L⁻¹ hydrocarbon concentration or 30 µg L⁻¹ WAF), only two species, *Fucus evanescens* (a kelp) and *Phymatolithon lenormandii* (encrusting coralline rhodophyte), declined in abundance (Bokn et al. 1993). On the other hand, in north-eastern India, Premila & Rao (1997) found that 0.02 ppm of crude oil was sufficient to inhibit growth of 10 species (one Phaeophyta, four Rhodophyta and five Chlorophyta) and that 72-hour LC50 values among these varied from 0.28 to 1.9 ppm, with green algae the most sensitive. As with seagrasses, the sensitivity of macroalgae to spills appears to decline from high- to low-intertidal levels on shores, with filamentous red algae the most susceptible (O'Brien & Dixon 1976). Subtidal macroalgae do not always die or exhibit reduced growth rates following oil spills (Peckol et al. 1990, Lewis & Pryor 2013), other than in the most severe situations (Jackson et al. 1989).

Reported toxic responses to oil have included a variety of physiological changes to enzyme systems, photosynthesis, respiration and nucleic acid synthesis (Lewis & Pryor 2013). Photosynthetic uptake has been the most commonly used physiological index in studies on the toxic effects of oil on algae (Lewis & Pryor 2013). Disruption of cellular membranes by certain types of hydrocarbons can have an adverse effect on photosynthesis (O'Brien & Dixon 1976). The double lipophilic layer is susceptible to swelling by hydrocarbons penetrating the outer protein layers. Darkening of cells and distortion of membranes has been reported by Kauss et al. (1973, cited in O'Brien & Dixon 1976), indicating possible interference with membrane structures (O'Brien & Dixon 1976). It has also been suggested that hydrocarbons dissolve in the lipid phase of the grana of chloroplasts (O'Brien & Dixon 1976). An increase in distance between individual chlorophyll molecules caused by membrane distortion and other disturbances to submicroscopic structures appears to impair photosynthetic ability (O'Brien & Dixon 1976).

Despite the well-established pool of literature on macroalgae exposure to petroleum oils, very few investigations have reported effects on species that are common in Australian waters (Lewis & Pryor 2013). Rather, most studies on macroalgae have occurred in response to northern hemisphere oil spills (Lewis & Pryor 2013), with a focus on the phaeophyte *Fucus*, a common temperate intertidal species. Noticeably absent from the literature on toxicity are the attached, meadow forming species within the *Sargassum* genus, which have widespread distribution across Australia's tropical north (especially in the Kimberley; Huisman et al. 2009) and also temperate south, and is thought to have in excess of 200 species (Edgar 2012). While no evidence indicates that *Sargassum* spp. and *Fucus* spp. respond similarly to hydrocarbon exposure, they do occur in the same taxonomic order (Fucales) and share similar morphologies, including flat, branched fronds that are either straplike or tapering, as well as characteristic vesicles. *Ulva* spp. and *Padina* spp., which belong to different taxonomic orders (Ulvales and Dictyotales, respectively) but also commonly occur in Australian waters, have both been reported with inhibited growth following oil exposure (Lewis & Pryor 2013).

Chronic and indirect effects of oil exposure

Impacts on reproduction and growth

Invertebrates Reduction in the success or extent of reproductive activity following exposure to oil has been shown for several different invertebrate species, indicating 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. Similarly, sublethal concentrations of crude oil were found to decrease brood numbers in amphipods when females were exposed during the incubation period (Linden 1976). Elmgren et al. (1983) also found that several months following the TSEIS oil spill in Sweden, female amphipods (*Monoporeia affinis* [= *Pontoporeia affinis*]) showed a significant increase in proportion of abnormal eggs.

Blumer et al. (1970) found that while an oil spill did not cause mortality, mussels failed to reproduce following the spill (with mussels from an area not affected by the oil spill reproducing normally). Other studies (Renzoni 1973, 1975; Nicol et al. 1977) showed that exposure to No. 2 fuel oil WAF affected sperm motility and reduced fertilisation in sand dollars and bivalves. Vashchenko (1980) reared sea urchin larvae from gametes obtained from adult *Mesocentrotus nudus* (= *Strongylocentrotus nudus*) maintained for 45 days in seawater containing 30 mg L⁻¹ of diesel. While larvae from control urchins developed normally, those produced from gametes of urchins maintained in the diesel-contaminated water had a higher proportion of abnormal and non-viable larvae after three 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 (*Metacarcinus magister* [= *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).

Decreased reproductive success of both brooding and broadcast spawning corals has been shown following exposure to oil. The hydrocarbon effects on adult colonies range from the premature expulsion of larvae (Cohen et al. 1977, Rinkevich & Loya 1977, Villanueva et al. 2011), significant reduction in the number of colonies with gonads in their polyps (Rinkevich & Loya 1977), and significantly smaller gonads (Guzmán & Holst 1993). Despite these effects, Negri & Heyward (2000) suggested that the early life stages of corals may in fact be more sensitive to hydrocarbons and showed that larval metamorphosis was inhibited by exposure to crude oil and dispersed oil. Epstein et al. (2000) exposed *Stylophora pistillata* larvae to Egyptian crude oil and reported no mortality from the WAF of the oil, no significant mortality from the dispersants (Emulgal C-100,

Biosolve, Inipol IP-90, Petrotech PTI-25, Bioreico R-93), but the dispersed oil fraction resulted in significant mortality. Furthermore, the authors reported a significant reduction in settlement rates from the WAF and dispersants, and no successful settlement was evident in the dispersed oil treatment. Anomalies in planula morphology and behaviour were also recorded in the dispersant and dispersed oil treatments (Epstein et al. 2000). Larval settlement and survival in *Porites astreoides* and *Orbicella faveolata* (= *Montastraea faveolata*) decreased with increasing concentrations of the WAF of DEEPWATER HORIZON crude oil, the dispersant Corexit 9500 and the dispersed oil (Goodbody-Gringley et al. 2013). Malampaya natural gas condensate has also been shown to result in larval mortality in *Seriatopora hystrix* and *S. guttata* (= *S. guttatus*) but not in *Stylophora pistillata*, *Pocillopora damicornis* and *P. verrucosa* (Villanueva et al. 2008).

There is also evidence that exposure to oil affects coral growth. Birkeland et al. (1976) exposed colonies of *Porites furcata* to Bunker C oil and reported a significant difference in growth between the control and treatment colonies. Guzmán et al. (1991) found a reduction in growth rates in three species (*Porites astreoides*, *Pseudodiploria strigosa* [= *Diploria strigosa*], *Orbicella annularis* [= *Montastraea annularis*]) following the major 1986 oil spill near Galeta in Panama, with growth rates of *Porites astreoides* being lower during the three years after the spill than before the spill (Guzmán et al. 1994). Following the Panama oil spill, *Siderastrea siderea* showed no reduction in growth rate (Guzmán et al. 1991), but the growth rate of *S. siderea* differed significantly before and after the spill (Guzmán et al. 1994). Decreased juvenile growth rates have also been reported in *Seriatopora hystrix*, *S. guttata* (= *S. guttatus*) and *Stylophora pistillata* following exposure to the WAF of Malampaya natural gas condensate (Villanueva et al. 2008).

Seagrasses While considerable variation in reproductive strategy occurs at the species level, perennial seagrasses typically rely on lateral shoot growth for local persistence and expansion, with use of seeds a secondary strategy (Larkum & den Hartog 1989). However, production of new plants from dispersed seeds can provide an important auxiliary mechanism for re-establishing meadows (Orth et al. 1994). While it could be presumed that responses to oil exposure among seagrasses with different life cycles and reproductive strategies will vary, there have been no systematic studies to specifically test if such differences occur. From the few reported studies available, species that rely heavily on recruitment from seeds appear to suffer greater impacts from residual oil in sediments than species with vegetative reproduction (Dean et al. 1998). Dean et al. (1998) reported that seeds collected from an oiled site had a higher germination rate than seeds from its paired reference site, but the seedlings produced from the oiled site also had higher rates of mitotic abnormalities.

Oiling also appears to affect flowering success (Houghton et al. 1993, Dean et al. 1998). In an assessment of seagrass health following the EXXON VALDEZ crude oil spill in Alaska, Dean et al. (1998) reported that the density of flowering shoots was more than twice as high at reference sites compared to oiled sites. The paucity of flowering shoots at oiled sites translated directly to a lack of inflorescences produced. These results were comparable to those of Houghton et al. (1993), who independently conducted surveys of eelgrass following the EXXON VALDEZ oil spill. It is difficult to predict the implications of a reduction in flowering success associated in oil spills for long-term viability of seagrass meadows, as in both instances, declines in flowering success were short-term (natural flowering success had recovered within two years post spill) and did not lead to reductions in shoot density in subsequent years (Houghton et al. 1993, Dean et al. 1998).

There has been no examination of the effect of oils on the various life stages of seagrasses, and tolerances of young versus mature plants are unknown.

Macroalgae Macroalgal reproduction varies considerably among phyla and species but, in general, involves mass production of gametes, zygotes and spores forming propagules that can be readily dispersed to nearby habitats. Gametes can be either male or female and are released by the development of reproductive bodies on the plant (Holmquist 1997, Peterson et al. 2002).

Once released, spores or eggs attach to a surface or substrate and then are fertilised and mature into zygotes, juvenile and eventually adult plants. Sensitivities to toxicants at different macroalgal life stages is common, and gametes, larva and zygote stages have all proven more responsive to oil exposure than adult growth stages (Thursby et al. 1985, Lewis & Pryor 2013). Of the response parameters reported by Lewis & Pryor (2013), germination, reproduction and growth rate have been more sensitive than other indicators of macroalgal health, with inhibition the dominant effect.

Impacts on movement, attachment and feeding of invertebrates

Animals not exposed to oil at concentrations high enough to kill them may still suffer mortality not directly related to the oil impact. For example, a reduced ability to move away from oiled areas or escape predators may reduce survival. Percy & 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 remained attached to the reef, but their grip was weakened. This may have affected their ability to survive strong breaking waves or resist predators. Similarly, in an experiment in the Arctic, 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.

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 seastar *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 seastars had been exposed to a very dilute (0.12 ppm) crude oil seawater mixture (O'Clair & Rice 1985). Corals have also been shown to respond to oil pollution by abnormal feeding reactions such as mouth opening or reduced pulsation rates (Reimer 1975, Cohen et al. 1977, Rinkevich & Loya 1979).

Diseases and physiological responses

A range of tumour and blood-type diseases of crustaceans and molluscs have been reported in animals exposed to oil spills (see Hodgins et al. 1977 for review of oil-induced disease responses in fish and invertebrates). Deoxyribonucleic acid (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 oil spill off Spain from the ship AEGEAN SEA (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 fishes than invertebrates (mussels and sponges) (Harvey et al. 1999).

Changes in the cellular physiological condition of coral have been shown following exposure to oil. Downs et al. (2006) measured the effects of a spill of intermediate grade oil on *Porites lobata*. Samples were collected three months after initial exposure. Changes were recorded at the cellular level in terms of lesions found adducted to coral macromolecules. The cellular physiology was consistent with the pathological profile that results from the interaction of corals with PAHs (e.g. an injury resulting from exposure to a xenobiotic). Similar results were recorded by Rougee et al. (2006) when exposing *Pocillopora damicornis* in the laboratory to fuel oil. Moreover, following a major oil spill in Panama, Burns & Knap (1989) found that corals stressed by hydrocarbons had altered protein-to-lipid ratios, suggesting a disruption in the lipid synthesis system. Coral mucus is rich in wax esters, triglycerides and other lipids and is an energy-rich link in the coral reef food chain (Benson & Muscatine 1974). As such, in addition to the adverse change in coral physiology, the disruption in the lipid synthesis system has the potential to cascade to other components in the ecosystem that are dependent on the corals for food and substrate (Burns & Knap 1989).

A concentration dependent change in the photosynthetic rate of the zooxanthellae was reported in *Stylophora pistillata* following exposure to the water-soluble fraction (WSF) of Iranian crude oil (Rinkevich & Loya 1983). Cook & Knap (1983) exposed colonies of *Pseudodiploria strigosa* (= *Diploria strigosa*) to varying concentrations (18–20 ppm) of Arabian light crude oil and the dispersant Corexit 9527, and while the hydrocarbon exposure alone did not affect carbon fixation, the dispersed oil resulted in significant impacts to photosynthesis. Knap et al. (1985) noted these effects were temporary however, and there was no evidence of a long-term effect on the coral skeleton. Similarly, Dodge et al. (1985) found no detrimental effects on calcification rates of *P. strigosa* when exposed to the same experimental conditions as that applied by Cook & Knap (1983) (i.e. Arabian light crude/Corexit 9527). There is also some evidence that at low levels of exposure to oil a slight enhancement of photosynthesis and calcification rates can occur (Rinkevich & Loya 1983, Dodge et al. 1985) even where it is suppressed at higher hydrocarbon concentrations (Rinkevich & Loya 1983). This stimulatory effect in response to low levels of pollutants is known as hormesis (Dodge et al. 1985). Jones & Heyward (2003) reported a reduction in the photochemical efficiency of zooxanthellae in *Plesiastrea versipora* exposed to produced formation water (3%–50% w/v). Consequently, it appears that sublethal exposure to hydrocarbons can have an adverse effect on the photosynthetic efficiency of zooxanthellae, but that the effect can be inconsistent and reversible.

Corals secrete mucus as a protective mechanism, and research has shown that its production varies with exposure to oil. Peters et al. (1981) exposed *Manicina areolata* colonies to No. 2 fuel oil for three months and noted an increase in mucous sensory cell activity after two, four and six weeks. Coral mucus, therefore, not only serves to protect coral tissue from oil but also as a mechanism of clearing oil from contaminated corals. Production of mucus may also serve as a major pathway of energy loss, given reports that 40% of the primary production in *Acropora* is lost as mucus (Loya & Rinkevich 1980). In stressed corals, this energy loss may constitute a large energy drain, which could further compromise the health of the colonies. Moreover, coral mucus serves as a food source to other reef organisms (Cole et al. 2008, Rotjan & Lewis 2008), and in the event of oil exposure, the mucus may therefore serve as a pathway for contamination (Shigenaka 2001).

Stimulation of epiphytic algal growth

The effects of oil on epiphytic micro- and macroalgal communities are complex. Petroleum oils are both toxic and sources of labile carbon for energy and growth (Altenburger et al. 2004). In theory, any stimulatory effects on growth of epiphytic algae on seagrasses can lead to indirect seagrass loss (Cambridge et al. 2007). If enhanced epiphytic growth covers the greater portion of a seagrass leaf surface, insufficient light becomes available to enable photosynthesis (Masini & Manning 1997) and, if persistent, can lead to declines in seagrass shoot density and ultimately seagrass loss (Cambridge et al. 2007). den Hartog (1986) suggested that the impact of reduced light as a result of enhanced epiphytic or phytoplankton loading could potentially be greater than the impact of the petrochemical toxicity on seagrasses, while Jackson et al. (1989) reported high levels of epiphytic algal growth on subtidal seagrass (*Thalassia testudinum*) for several months following the 1986 oil spill at Galeta in Panama. The level of epiphytic growth in the Galeta example, however, was insufficient to induce changes in seagrass health (Jackson et al. 1989). Despite the potential for this indirect cause–effect pathway, there is limited evidence to confirm the seriousness of this effect. Further, assuming there is a legitimate link, there is no evidence to confirm that the level of stimulation would result in the same magnitude of epiphytic growth (and loss of seagrass) induced by other nutrient sources (Cambridge et al. 2007).

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). The 1967 TORREY CANYON spill resulted in extensive mortality to herbivorous limpets and other invertebrates on the Cornwall coast in the United Kingdom when 40,000 tonnes of Kuwait crude oil washed ashore. The bare space was rapidly colonised by ephemeral chlorophytes (*Blidingia* and *Ulva* [= *Enteromorpha*]) and then dense stands of the phaeophyte *Fucus*, which resulted in the death of most remaining barnacles. Limpet recruitment then began to occur beneath the plant canopy and displaced the *Fucus* after about five years, but the species composition, biomass and abundance was greatly different and fluctuated dramatically for several years, and the shoreline did not begin to resemble pre-spill characteristics until the 1980s (Hawkins & Southward 1992, Hawkins et al. 2017a). Following the 1993 BRAER oil spill in Scotland, Newey & 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 were 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).

Oil spills have the potential to have community-scale effects. Despite this, most studies on the topic tend to track impacts on singular species over relatively short temporal scales. Peterson et al. (2003), however, suggest that expectations of short-term singular species effects from oiling ignore the complexity of ecological interactions and should be replaced by a better understanding of the sequence of delayed indirect effects over much longer timescales (Peterson 2001). Changes in community composition in macroalgal communities following the 1989 EXXON VALDEZ incident provides an interesting case in point. Following this oil spill event, there was considerable loss of macroalgal cover (*Fucus distichus* [= *Fucus gardneri*]) within the intertidal zone, which triggered a cascade of indirect effects (Peterson et al. 2003). Freeing of space on the rocks and the losses of important grazing (limpets and periwinkles) and predatory (whelks) gastropods combined to promote initial blooms of ephemeral green algae in 1989 and 1990 and an opportunistic barnacle (*Chthamalus dalli*) in 1991 (Peterson et al. 2003). Absence of structural algal canopy also led to declines in associated invertebrates and inhibited recovery of *Fucus distichus* itself, whose recruits survive better under the protective cover of the adult plants. Those *F. distichus* plants that subsequently settled on *Chthamalus dalli* became dislodged during storms because of the structural instability of the attachment of this opportunistic barnacle (Peterson et al. 2003). After apparent recovery of *Fucus distichus*, previously oiled shores exhibited another mortality event in 1994, which was suggested to be a cyclic instability associated with simultaneous senility of a single-aged stand (Driskell et al. 2001).

The 1986 Galeta oil spill in Panama provides another interesting case study of selective effects at the community scale. Seagrasses and associated algal epiphytes provide important feeding, nursery and refuge habitats for marine fauna (Bostrom & Bonsdorff 2000, Smit et al. 2005). The Galeta spill resulted in large declines in seagrass (*Thalassia testudinum*) in intertidal areas, although subtidal seagrass meadows remained, albeit largely covered in epiphytic algae (Jackson et al. 1989). Within the dead but intact intertidal seagrass root-rhizome mats, Jackson et al. (1989) reported losses in some macroinvertebrate fauna (including amphipods, tanaidaceans and ophiuroids), while relative abundances of other macroinvertebrate taxa (bivalves, gastropods and polychaetes) were consistent with surrounding unoiled seagrasses.

The selective nature of oil toxicity caused by the resilience of some groups can, in turn, also alter community structure. Following a large 2006 oil spill off Estonia, Kotta et al. (2008) compared the

abundance of guilds of herbivores, suspension feeders 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 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 six years.

Peterson (2001) and Peterson et al. (2003) documented the long-term 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. 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 (*Leukoma staminea* [= *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. However, Payne et al. (2008) argued (on the basis of sampling for 11 years longer than Carls et al. 2001) that levels of contamination in mussels had reduced to very low levels by 2006, and that bioaccumulation by mussels only presented a problem when substrate with adsorbed oil was disturbed. 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).

While petroleum hydrocarbon uptake and accumulation can occur in seagrasses and macroalgae, uptake does not appear to be a major entry point into food chains (Lewis & Pryor 2013), although there remains some research interest in the topic. For example, after the NASSIA tanker accident (14 March 1994), there were elevated concentrations of oil (total petroleum hydrocarbons [TPHs] up to 196.76 $\mu\text{g g}^{-1}$) reported in five different species of marine macroalgae occurring in the Bosphorus Straits, Turkey (Binark et al. 2000). There is also evidence that the bioaccumulation rate is likely to be species specific (Getter et al. 1985, Binark et al. 2000), with uptake also shown to be affected by salinity and temperature (Wolfe et al. 1998b). The influence of dispersants on uptake of some fractions of petroleum hydrocarbons (e.g. [^{14}C]naphthalene) has ranged from a limited effect to as much as a 10-fold increase for some marine algae (Wolfe et al. 1998a) and seagrasses (Lewis & Pryor 2013).

The toxic effects of tissue-accumulated hydrocarbon concentrations, such as those referred to previously, are almost unknown (Lewis & Pryor 2013). In laboratory experiments undertaken by Navas et al. (2006), *Daphnia magna* were fed with algae previously exposed to fuel, but no toxic effects were detected despite the presence of chemicals (elevated ethoxyresorufin-O-deethylase (EROD) activities of up to 16 pMol $\text{mg}^{-1} \text{min}^{-1}$ in RTG-2 cells) that could cause sublethal effects to organisms (Navas et al. 2006). The scarcity of threshold effect data, in general, reduces the value of most oil-related bioaccumulation studies to the risk assessment process and prevents the calculation of tissue screening concentrations and toxic units that have proven useful for non-oil contaminants and aquatic biota (Shephard 1998).

In addition to bioaccumulation, novel pathways of injury to marine fauna from offshore oil spills have also recently been reported following the DEEPWATER HORIZON spill (Powers et al. 2013). The pelagic brown alga *Sargassum* forms a floating platform, or mat, which supports biodiversity

and secondary production. According to Powers et al. (2013), the vast pool of oil resulting from the DEEPWATER HORIZON oil spill came into contact with a large portion of the Gulf of Mexico's floating *Sargassum* mats, leading to three pathways for oil spill-related injury: (1) *Sargassum* accumulated oil on the surface, exposing animals to high concentrations of contaminants; (2) application of dispersant sank *Sargassum*, thus removing the floating mat habitat and potentially transporting oil and dispersant vertically; and (3) low oxygen surrounded the floating mat habitat, potentially stressing animals that reside in the alga. Such pathways represent direct, sublethal and indirect effects of oil and dispersant release that minimise the ecosystem services provided by floating *Sargassum* (Powers et al. 2013).

Loss of genetic diversity

Loss of genetic diversity in the razor clam *Ensis siliqua* four years after the 2002 PRESTIGE oil spill in Spain was attributed to this event (Fernandez-Tajes et al. 2012). These authors compared prespill data of genetic diversity with that from the population that recovered following the oil spill. The latter had been produced from a very much smaller population than existed before the oil 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 is 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, although this did not occur in this case, at least in the initial years following the spill.

Factors influencing impacts

Impacts of residual oil

One important mechanism that prolongs the effects of an oil spill occurs when oil is sedimented into the substrate. Oil is readily incorporated 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 continued to leach from mangrove sediments and to bioaccumulate in bivalve molluscs. Similarly, more than seven years after a long history of acute and chronic oil spill pollution in Curacao, coarse rubble habitats cemented together with tar had a lower abundance and diversity of gastropod molluscs than unpolluted areas of similar habitat (Nagelkerken & Debrot 1995).

Sedimented 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 spill from a storage facility in Long Cove, Maine, USA, as new recruits burrowed into sediment contaminated to 250 ppm oil at 15–25 cm below the surface. Gilfillan & 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.

Hydrocarbons can adsorb on aquatic plant-rooted soils and sediments where they can persist for up to 30 years (Lewis & Pryor 2013). Research has demonstrated that substrate oiling has adversely

impacted other aquatic primary producers such as freshwater phytoplankton, wetland plants (Gilfillan et al. 1989, Naidoo et al. 2010) and mangroves (Klekowski et al. 1994). Phytotoxicities of substrate-bound hydrocarbons to seagrasses and macroalgae are important to understand, since the protection of below-ground root-rhizome systems are likely to be important for plant regeneration after oil spills. It is also possible that smaller ephemeral seagrasses, which tend to have shorter rhizomes that persist for weeks to months, may be differentially affected compared to larger seagrasses, which tend to have longer and more persistent rhizomes that exist for months to years. Furthermore, evaluation of above-substrate toxic effects may underestimate below-substrate toxic effects (Lin et al. 2002).

Composition of oil and weathering

Refined oils, diesel and heavy bunker fuel oils are apparently more toxic than crude oil. Anderson et al. (1974) compared the toxicity of a heavy fuel oil (Bunker C), a light fuel oil (No. 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), 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 hydrocarbons. There has been very little work done specifically on the toxicity of natural gas condensates (except see Negri et al. 2015) that are particularly relevant to Australia's North West Shelf, 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 and other properties. 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 acute toxic impacts than freshly spilled oil. Apart from the oil's composition (or type), the main factors that will influence weathering are wave mixing, which will be affected by wind speed and temperature which effects of rates of evaporation and biodegradation (Neff et al. 2000, Venosa & Zhu 2003, Lee et al. 2013).

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 predistilled 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 percentage 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 percentage of the WAF that resulted in abnormal development on 50% of the larvae after 60 hours (60-hour EC50). These varied between 11%–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.

Use of dispersants and shoreline clean-up

The use of chemical dispersants in oil spill clean-up offers the benefit of reducing the threat of oil reaching shorelines and intertidal communities. However, 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 to marine invertebrates than the oil on its own. 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 (Smith 1968, Southward & Southward 1978, Hawkins & Southward 1992), with sites where dispersant was not used recovering after three years compared to as much as 15 years at sites where dispersant was used (Hawkins et al. 2017a,b). It should be stated, however, that modern dispersants are less toxic (Hawkins et al. 2017a), undergo more rigorous toxicity testing and are subject to an approvals process not in place in 1967 when first-generation dispersants were used. 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 10 times more toxic to embryos of grass shrimp (*Palaemon pugio* [= *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. 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 invertebrates than oil–water mixtures alone has led to calls for oil spill clean-ups to employ burning off the volatile fraction instead of using dispersants (Georgiades et al. 2003).

The situation with seagrass and macroalgae and dispersants is far less clear, and conflicting findings have complicated the decision-making process of whether or not to disperse oil (Wilson & Ralph 2012, Lewis & Pryor 2013). Research outcomes include dispersed oil posing a greater threat than non-dispersed oil, dispersed oil posing less of a threat than non-dispersed oil, and that neither oil nor dispersed oil impact seagrass or macroalgae (Wilson & Ralph 2012, Lewis & Pryor 2013). Further complicating the matter, responses to dispersed oils can be highly variable between species (Thorhaug et al. 1986).

Seagrasses have been shown to absorb more aliphatic and aromatic oil fractions when the oil is dispersed, therefore increasing its toxicity (den Hartog 1986). Dispersants are thought to affect the waxy cuticle of the seagrass blade and, in doing so, increase the penetrability of the dispersed oil to the photosynthetic organs, particularly the thylakoid membrane (Wolfe et al. 1998a). Most commonly, it appears that non-dispersed oil leads to less photosynthetic stress compared with the addition of a chemical dispersant. Wilson & Ralph (2012) and Hatcher & Larkum (1982) both found that dispersed oil had a greater impact on seagrasses than untreated oil. However, Ralph & Burchett (1998) found no comparative difference in photosynthetic activity of *Halophila ovalis* when exposed to crude oil and dispersed crude oil (both treatments induced similar declines). Macinnis-Ng & Ralph (2003) also found little difference between the impacts of oil and dispersed oil on *Zostera capricorni* in field experiments (but showed a greater impact from oil compared to dispersed oil in laboratory experiments).

It is also important to consider the type of dispersant used. Wilson & Ralph (2012) found that the toxicity of dispersed crude oil to *Z. capricorni* was specifically related to the brand of dispersant and the petrochemical loading in the water column. While both Corexit 9527 and Ardrex 6120 dispersed oil treatments produced negative impacts to *Z. capricorni*, the Ardrex 6120 dispersed treatment had a more sustained negative impact on the seagrass. For *Halophila ovalis*, $\Delta F/F'_m$ impacts were

detected within the first four hours but were followed by full recovery. These laboratory experiments suggest that dispersed crude oil is more toxic to *Zostera capricorni* and *Halophila ovalis* than non-dispersed crude oil, and that Ardrex 6120 dispersed crude oil is slightly more toxic than Corexit 9527 dispersed crude oil.

Although the increased toxicity of dispersed oils has been demonstrated in numerous studies, the practise may be warranted in situations offshore, where reduced shoreline oiling and reduced risk of sedimented oil 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.

Mechanical cleaning of oil-impacted shorelines has also been shown to be potentially more destructive than the effects of the oil spill on its own (Broman et al. 1983). Rolan & Gallagher (1991) reported that even more than eight years after the 1978 Esso BERNICIA oil spill in the Shetland Islands, Scotland, mechanically cleaned shorelines (i.e. by removal of oiled substrate) had not recovered, whereas those areas that were inaccessible to mechanical cleaning had recovered within one year despite the persistence of some residual oil (Rolan & Gallagher 1991). After the 2002 PRESTIGE oil spill in France, Castège et al. (2014) attributed high-pressure hot-water cleaning as one reason why communities did not recover normally, including the permanent loss of some sponges (e.g. *Hymeniacidon perlevis* and *Tethya* sp.). Le Hir & Hily (2002) also found that recovery of the rocky shore after the 1999 ERIKA spill in France was retarded by high-pressure cleaning as it removed macroalgae that had survived the spill, thereby altering the succession dynamics and recovery of grazers to the area. In a review of the recovery periods described for 34 oil spills, Sell et al. (1995) concluded that clean-up and treatment usually resulted in longer recovery periods. Hawkins & Southward (1992) stated that one of the most important lessons from the 1967 TORREY CANYON spill was that dispersants should not be used against spills impacting exposed rocky shorelines (see also Hawkins et al. 2017a).

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 considered 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°C to 16.5°C, and then to 31.2°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⁻¹ to 13 mg L⁻¹ to 4 mg L⁻¹ at 8.5°C, 16.5°C and 31.2°C, respectively. Consistent with this, Fisher & Foss (1993) compared toxicity of fuel oil to embryos of grass shrimp *Palaemon pugio* (= *Palaemonetes pugio*) at different temperatures and found that the effects of toxicity onset earlier at higher temperatures.

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 effects of sedimented oil, resupply of disturbed sediments 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 (Edgar & Barrett 2000, Loughheed et al. 2002, Edgar et al. 2003) for shipping accidents in Australia and the Galapagos. Nevertheless, exceptions exist, 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 previously 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 oil spill in the Baltic Sea off Estonia. On Macquarie Island, invertebrate abundances at impacted sites on exposed shores had returned to pre-spill impact levels and were comparable to control sites after seven years, but this was not the case among the more sheltered kelp holdfast habitat, which had not recovered after that time (Smith & Simpson 1995).

Several 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, observed differences in the field following oil spills are generally due more to zonation of intertidal animals rather than differential sensitivities. 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 dependent 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 & Burns 1977), and the 1986 refinery spill at Galeta, Panama (Burns et al. 1993). Incorporation of oil into intertidal habitats prolongs the exposure of

animals to residual oil, both through burrow activities and as the oil is re-exposed due to erosion or other disturbances. 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. (2011) found greater impacts lower on the shore on beaches, rather than on the upper shore. For the same spill, impacts were 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 all crabs present along hundreds of kilometres of coastline were inferred to have been 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 DEEPWATER 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 nutrient enrichment after the spill and subsequent increased bacterial production). After the 2002 PRESTIGE oil spill off Spain, Sanchez et al. (2006) surveyed depths of 70–500 m where tar aggregates 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.

Capacity for recovery

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 three years for rocky shores and five years for salt marshes. However, numerous exceptions were identified. 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 was well advanced by two 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 clean-up had not been carried out. Mostly these sites were in a recovering stage two to three years post spill, except in three cases where they were still in the recovery phase 10 years post spill. Sell et al. (1995) noted that in at least one of these cases, the clean-up 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 et al. (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. Moore (2006) also reviewed the impacts of several oil spills and concluded that where oil had been removed from the environment and the clean-up process had not resulted in physical damage to biota that recovery was generally rapid, in the order of a few years, but that recovery was much longer when long-term oil contamination remained present.

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 TSEIS oil spill, *Pontoporeia* amphipod numbers were still depressed, while the polychaete *Bylgides sarsi* (= *Harmothoe sarsi*) had returned to preimpact densities, and *Limecola balthica* (= *Macoma balthica*), which had not been affected by the spill, became significantly more abundant. Elmgren et al. (1983) estimated it would take 5–10 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 monitored impact and control sites, and after three years, recruitment in the spill areas remained reduced relative to unimpacted sites; some species of littorinids had not re-established after three years. Garrity & Levings (1990) estimated it would take 5–10 years (or perhaps longer) for the relative densities of species in the affected area to return to normal. After the 1999 ERIKA spill in France resulted in 100% mortality of the sea urchins *Paracentrotus lividus* and *Psammechinus miliaris*, Barillé-Boyer et al. (2004) found it took two years until the first urchins reappeared and three years for densities to return to preimpact levels ($\sim 60 \text{ m}^{-2}$).

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. One large-scale (900 m²) simulated oil spill in Panama caused total mortality of the sea urchins present, and those that reappeared seven months later (as recovery started to take place) were smaller and most likely recruits rather than migrants (Ballou et al. 1989). Egres et al. (2012) experimentally oiled sandy intertidal sand flat plots with diesel in Brazil 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 m², each oiled with 2.5 L) and then rapid recovery of the treated plots within two days, resulting from immigration by animals from outside the treated 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 patchy at scales of metres, and not heavy nor persistent enough to be incorporated into sedimentary processes, recolonisation of soft-substrate habitats can occur rapidly.

For seagrass or macroalgal habitat subject to only minor disturbance from hydrocarbon exposure, the duration of recovery can be rapid—within months to less than a year (Dean et al. 1998). Following the 1989 EXXON VALDEZ oil spill, eelgrass (*Zostera marina*) beds in heavily oiled bays were exposed to moderate concentrations of hydrocarbons. A year after the spill, Dean et al. (1998) quantified injuries (shoot density and flowering shoots) to eelgrass to be slight and not significantly detectable relative to undisturbed sites (noting the power of the experimental design was low and may have masked statistical outcomes). Five years after the EXXON VALDEZ oil spill, follow-up surveys indicated no significant differences between oiled and reference sites (measured via biomass, seed density, seed germination or the incidence of normal mitosis in seedlings), and no signs of the elimination of eelgrass beds (Dean et al. 1998). Dean et al. (1998) concluded that the eelgrass populations recovered from possible damage by 1991, which coincided with sharp declines in hydrocarbon concentrations.

Evidence for long-term recovery (>5 years)

Despite some studies finding good recovery after periods of three to six years, others have demonstrated incomplete recovery or lagging sublethal effects that persisted for decades. Long-term monitoring to 1990 following the 1967 TORREY CANYON spill indicated that recovery of communities, to the extent that they reflect prespill species assemblages and their natural levels of spatial and temporal variability, takes close to 15 years (Hawkins & Southward 1992). 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 & Burns (1977) found significant effects of the FLORIDA barge oil spill, which affected salt marsh habitat in West Falmouth, Massachusetts, USA, remained after seven years with fiddler crabs (*Minuca pugnax* [= *Uca pugnax*]) at reduced densities compared to prespill 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 reoccurring. Another survey 30 years after the event (Reddy et al. 2002) found similar results, with oil still present in cores between 6–28 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 & 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 six 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 eight to nine months. Despite this incident, the intertidal gastropod *Littorina keenae* (= *Littorina planaxis*) survived, but subtidal gastropods including abalone (*Haliotis* spp.) were killed, with reduced numbers seven 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 three to six generations (5–10 years in the case of bivalves) before populations recovered fully. In a longer-term investigation, Dauvin & Gentil (1990) found that populations of most amphipod species had recovered after 10 years to densities similar to before the AMOCO CADIZ spill; however, some formerly abundant species were not found in any post-spill surveys and had probably been extirpated. After the 2002 PRESTIGE oil spill in France, Castège 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.

There are limited studies on the recovery of coral communities following oils spills. Guzmán et al. (1994) reported that there was no evidence of coral recovery five years after the 1986 major oil spill in Panama. An understanding of the capacity and timeline to recovery can be drawn from other impacts to coral reefs such as coral bleaching from elevated sea surface temperatures and physical damage from cyclones. Studies have shown significant recovery of corals from these types of impacts in less than 15 years in some cases, longer in others (Baker et al. 2008). Recovery may be delayed if there is residual oil in the sediment, especially where PAHs continue to be released.

Where exposure to oil has resulted in severe impacts to a seagrass or macroalgal habitat (e.g. Galeta oil spill in Panama), there has been little follow-up reporting on recovery. This has considerably limited our understanding of the possible capacity, or required durations, of seagrass and macroalgal assemblages to return to predisturbance conditions. Based on other investigations of where seagrass meadows have been severely disturbed (i.e. considerable reductions in shoot density and loss of leaf biomass), recovery may not commence for many years after the stressor has ceased (McMahon et al. 2011).

In terms of community-scale impacts (i.e. those involving both the plant assemblages and their associated fauna), the persistence of toxic subsurface oil and chronic exposures in sediments (even at sublethal levels) can result in delayed population reductions and cascades of indirect effects that can significantly hamper ecosystem recovery for many years (Peterson et al. 2003). Peterson et al. (2003) reported chronic exposures to *Fucus distichus* (= *Fucus gardneri*) assemblages after the EXXON VALDEZ spill, which led to a complex pattern of shifts in community composition represented by opportunistic species, many of which did not perform the same ecosystem functions as the original community.

Assessing impact and recovery from oil spills

A number of excellent best practice guidelines for oil spill response and monitoring have been published recently in Australia (Hook et al. 2016) and internationally (IPIECA-IOGP 2015, 2016); however, a number of key lessons learned are apparent from the literature reviewed here and are worth emphasising when assessing the impacts of oil spills on marine animals and plants.

Use best practice study design

Best practice is a before-after-control-impact (BACI) with well-established time series of baseline data (Green 2005) or 'beyond' BACI, which extends the method to multiple control sites and temporal replication of preimpact sampling (Underwood 1991, 1994). Good examples in the Australian context are Edgar & Barrett (2000) and MacFarlane & Burchett (2003), although often the availability of preimpact 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 prespill baseline data existed in impacted and unimpacted areas. Ideally, this should include multiple impacted and non-impacted sampling locations (e.g. Southward & Southward, 1978). Most often this was fortuitous, such as the overlap of high human population density, regional research laboratories and shipping lanes that exists in Europe. Nevertheless, availability of baseline data does not necessarily involve serendipity. Quantitative baseline surveys encompassing thousands

of species using the Edgar & Barrett (2000) method have now been undertaken at over 2000 sites around the Australian continent through the Reef Life Survey programme, including Vulcan Shoal and >100 other locations on the North-West Shelf (Edgar & Stuart-Smith 2014, Edgar et al. 2016). As a consequence, future spills affecting shallow reef systems in Australia have a high probability of prespill abundance data for hundreds of species in the near vicinity, plus comparable reference sites enabling a BACI statistical design, while also providing a monitoring framework for continental-scale state-of-the-environment reporting (Stuart-Smith et al. 2017).

Other planned approaches involve establishment of baselines in areas of intensive or frequent industrial use, as well as control sites, as occurred with the long-term seagrass monitoring programme in Gladstone. This was established several years before the 2006 GLOBAL PEACE spill in Gladstone and subsequently provided an excellent prespill baseline (Taylor et al. 2006, 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). Smith & Simpson (1995, 1998) and Schlacher et al. (2011) 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 (Johnson 1977, Peterson 2001), 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 or even annually over 50 years (e.g. Hawkins et al. 2017a), but generally these were in the worst cases where recovery may never be complete and damage perhaps permanent (e.g. Teal et al. 1992, Reddy et al. 2002 in the case of the 1969 West Falmouth, USA spill) or where interactions of recovery with other impacts such as imposex from tributyl-tin anti-fouling paints and climate-driven fluctuations and recent warming had become the focus of the research (Southward et al. 2004, Hawkins et al. 2017a,b). 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 programme (such as occurred with the 1989 EXXON VALDEZ spill; Payne et al. 2008) can confuse interpretation and contribute to debate about impact and recovery.

Consider a wide range of taxa

Different species have varying sensitivities to impact, and with the limited state of knowledge, it is currently difficult to pick winners in terms of the best sentinel species. Unpredictable impacts involving species interactions are likely in the most severe spills and may otherwise go unnoticed.

Conclusions

Marine invertebrates, algae and seagrasses are sensitive to the toxic effects of oil. Depending on the intensity, duration and circumstances of the exposure, invertebrates 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. Although crude oil exposure to seagrasses and macroalgae can be toxic, toxicity varies among species, and, in general, marine algae and seagrasses are less sensitive to oil than marine invertebrates, with generally faster recovery rates. Some commonly occurring species of seagrasses in Australia may be less sensitive than those that occur elsewhere based on the studies reviewed.

While a range of generalities can be stated about the response of marine invertebrates, algae and seagrasses 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 clean-up 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 blowout. In Australia, crude oils from different oil fields possess a wide range of densities.

The degree of exposure 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, the extent of dilution 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 mechanical cleaning of oil from the substrate resulted in higher mortality and longer recovery times for marine invertebrates than the oil on its own. Although this was also true for the use of dispersants in the nearshore environment, their role in achieving dilution and preventing excessive oil stranding leading to chronic exposure to sedimented oil probably weighs in favour of their use where sedimentary habitats are threatened. For exposed rocky shores, the evidence weighs in favour of not using dispersants.

Likewise, for seagrass and macroalgae, the evidence suggests chemically dispersed oils are more toxic than crude oils, although results differ between studies and are not as clear-cut as for marine invertebrates. Research outcomes range from dispersed oil posing a greater threat than non-dispersed oil, to dispersed oil posing less of a threat than non-dispersed oil, to neither oil nor dispersed oil impacting seagrass or macroalgae. Further complicating the matter, responses to dispersed oils can be highly variable between species. As dispersed oils are significantly more toxic than oil alone, the use of dispersants offshore to reduce sea surface oil, improve dilution, aid weathering and decrease concentrations before shorelines are impacted should be considered on a case-by-case basis and is likely to be warranted in some situations, especially in Australia where a large number of offshore installations exist.

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 and plants 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 effects of sedimented oil which is re-supplied as sediments are subsequently 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 here 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 such as cerithiid gastropods (creepers) and some nematodes and polychaetes appear more 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 such as amphipods and isopods, gastropod molluscs and echinoderms. Uncertainty exists over the sensitivity of sponges with the very few studies that have been undertaken giving conflicting results. In tropical areas, scleractinian and hydrozoan corals are affected along with anemones and zoanths. Macroalgae are also susceptible to direct contact, while seagrasses have greater resilience due to rhizomes below the sediment surface. 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 severe spills of oil from ships or refineries close to shore. There have been no major impacts to subtidal reefs detected from oil spills in Australia to date.

The extensive literature on the effects of oil on coral shows that the potential damage to coral reefs is of particular concern, and there is substantial evidence that exposure to oil spills can be toxic to corals. Some corals are less resilient than others, and changes may occur to the gross percent cover of corals or the composition of different coral species. Branching coral (e.g. *Acropora* and *Pocillopora*) are generally more susceptible to impacts from oil than massive corals or corals with large polyps (e.g. *Favia* and *Fungia*). Studies have shown a range of impacts including changes in growth rate, feeding behaviour, cellular physiological condition, photosynthetic ability of zooxanthellae, fecundity, settlement, bioaccumulation and mortality. However, some studies also demonstrate recovery of corals from these effects over time. Toxicological and physiological studies are important to understand direct effects on corals in terms of growth and reproductive capacity. Corals are most vulnerable to oil spills during coral spawning, and subsequent larval and juvenile life stages, when they drift on or near the surface. Hydrocarbons exposed to ultraviolet radiation in

surface waters may photo-oxidise, which may increase the toxicity and bioavailability of hydrocarbon components such as PAHs and could impact the larval phase of corals.

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 powerful 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. Decapod crustaceans (crabs and shrimps) are another group that appear particularly vulnerable based on the studies reviewed. This was particularly true of burrowing species in sedimentary habitats, which can in turn drive oxygenation of shallow sediments and thus play a keystone role as habitat engineers.

Most of the available literature on the effects of oil on seagrasses and macroalgae show results of acute responses and/or acute exposure, which can be highly variable depending on the experimental set-up, oil type, and species assessed. Commonly reported acute toxic effects of oiling on seagrasses and macroalgae include reduced rates of photosynthesis and respiration, which can lead to morphological changes in the plant structure (loss of leaves/thalli, reduced biomass, etc.), and ultimately death. However, acute toxicity tests may not necessarily offer the best understanding of a hydrocarbon effect, and there is a need for more thorough assessment of long-term implications for oil spills in nearshore and intertidal marine 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. First, 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 occurs when oil is sedimented. Release from competition when formerly dominant taxa are removed by oiling may also contribute. Second, 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.

Although observations from field studies suggest that exposure to oil can affect reproductive success among seagrasses and macroalgae, there remains a large gap in the literature around this topic. Further, it is difficult to make generalisations as there is a considerable variation among different seagrasses, from large perennial species to small ephemeral species, and responses to different oil types are also likely to vary accordingly. To better understand the effects of hydrocarbons on the diversity of reproductive strategies and life-cycle stages, a more systematic approach to assessing these effects is required beyond the limited information presently available from field observations.

In terms of recovery, it appears that where oil exposure has led to minimal disturbance to a seagrass or macroalgal assemblage, the duration of recovery can be rapid—within months. However, where exposure to oil has resulted in severe impacts to seagrasses or macroalgae, there has been little follow-up reporting on recovery. This has limited our understanding of possible duration times required for seagrass and macroalgal assemblages to return to abundances/biomass similar to pre-disturbance conditions.

The key factor in all these considerations of oil impacts on marine animals and plants 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, seagrasses and

macroalgae 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 prespill 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, prespill 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 organisms, and to test the response of a range of receptor species 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 51 significant oil spills in Australia between 1970 and 2017. Of these, six occurred offshore with no likely or expected impact on benthic invertebrates or macrophytes. Despite the potential for oil spills to impact marine invertebrates, only 24 cases had potential direct effects of oil studied, and 19 cases had only cursory or no assessment of impact. Of those 24 spills where impact assessments are available in published or unpublished reports, only eight considered impacts on invertebrates, with many others focussing on birds or 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. Despite the majority of spills in Australia since 1970 being in temperate Australia, this is not the current trend. Ten of the 14 spills this century have been in the tropics, seven along the north Queensland coast in proximity to the Great Barrier Reef, and all four spills associated with offshore oil and gas infrastructure have been since 2009.

We also found very few assessments of the toxicity and sublethal effects of oil on Australian marine invertebrates and macroalgae, but that seagrasses were better studied. Studies undertaken have nearly all been confined to 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 marine plants and invertebrates may well be 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, echinoderms, and most macroalgae and tropical seagrasses. 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.

Acknowledgements

Much of the material examined in this review was assembled as part of a series of unpublished white papers prepared for the Australian Petroleum Production and Exploration Association (APPEA). Christina Street and Meryn Scott from CSIRO provided assistance with compiling the literature used in this review, and Christine Hanson from BMT assisted with editing. We thank Paul Irving from the Australian Maritime Safety Authority (AMSA) for providing access to AMSA records of recent pollution events between 2013 and 2015 and Jon Moore for providing helpful comments on the manuscript.

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