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Distinguishing globally-driven changes from regional- and local-scale impacts: the case for long-term and broad-scale studies of recovery from pollution

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Abstract

Marine ecosystems are subject to anthropogenic change at global, regional and local scales. Global drivers interact with regional- and local-scale impacts of both a chronic and acute nature. Natural fluctuations and those driven by climate change need to be understood to diagnose local- and regional-scale impacts, and to inform assessments of recovery. Three case studies are used to illustrate the need for long-term studies: (i) separation of the influence of fishing pressure from climate change on bottom fish in the English Channel; (ii) recovery of rocky shore assemblages from the *Torrey Canyon* oil spill in the southwest of England; (iii) interaction of climate change and chronic Tributyltin pollution affecting recovery of rocky shore populations following the *Torrey Canyon* oil spill. We emphasize that “baselines” or “reference states” are better viewed as envelopes that are dependent on the time window of observation. Recommendations are made for adaptive management in a rapidly changing world.

Keywords: Climate change, Long-term monitoring, Overfishing, Pollution, Torrey Canyon oil spill, Tributyltin (TBT)

1 Introduction

The world's oceans are rapidly changing. The climate of the Earth is changing, largely due to anthropogenic greenhouse gas emissions (see IPCC 2014 and references therein). Climate projections anticipate a warmer and more extreme world of droughts, floods, and stormier and rising seas (see IPCC 2014 for review). Dissolution of emitted carbon dioxide is also causing the pH of the oceans to decrease (Doney et al., 2009). Interacting with these global environmental changes are a variety of other anthropogenic impacts acting on marine ecosystems at a variety of spatial and temporal scales. These include global-scale impacts, such as removal of large widely-migrating long-lived apex pelagic predators (Estes et al., 2011; Jackson et al., 2001; Myers and Worm, 2003; Veit et al., 1997) and global homogenisation of marine biota through invasion of non-native species (Firth et al., 2016; Olden et al., 2004; Stachowicz et al., 2002b). Some pollutants also operate at global scales. For example, mercury can accumulate and bio-magnify up food webs to large pelagic species such as tuna (Cai et al., 2007; UNEP, 2013). Perfluorated compounds are another example of a globally pervasive pollutant (e.g. Ahrens et al., 2010). Pollution by plastic litter and its breakdown to micro-plastics can now also be considered a global issue (Andrady, 2011; Browne et al., 2011; Thompson et al., 2004).

Regional-scale impacts include overfishing of demersal (bottom) fish (Hutchings, 2000; Jennings and Kaiser, 1998; Tillin et al., 2006), eutrophication of semi-enclosed seas such as the Adriatic (Barmawidjaja et al., 1995; Crema et al., 1991) and Baltic (Bonsdorff et al., 1997; Suikkanen et al., 2007), and damming of major river systems such as the Yangtze (Jiao et al., 2007). There are pollutants that have been prevalent at the regional scale, such as Tributyltin (TBT) pollution that influenced extensive areas of coastline all over the world between the 1970s and 2000s (e.g. much of the south coast of England was affected due to multiple sources from adjacent marinas and ports; Bray et al., 2012; Bryan et al., 1986; Spence et al., 1990).

Many impacts occur at local scales, such as habitat loss and degradation due to urban developments for transport infrastructure, industrial plants, tourism and residential property (Firth et al., 2016). These impacts are usually the result of local decision-making by the public or private sector. Such impacts can, however, scale up to have regional-scale implications (Airoldi and Beck, 2007; Lotze et al., 2006). A classic example is the northern Adriatic coast where local municipalities have modified the coast to support tourism at the scale of less than a km. These interventions now occur almost continuously over 100 s of km of coastline, with shore-parallel and -perpendicular sea defences, often coupled with beach replenishment schemes (Airoldi et al., 2005).

In this perspective article, we examine how broad-scale climatic fluctuations and more recent global climate change interact with regional- and local-scale impacts on the marine environment (see also Hawkins, 2012). We start by examining the interaction of climate fluctuations with regional-scale fishing pressure, taking advantage of observations of fish populations in the English Channel stretching back over 100 years. We then consider the interaction of climate fluctuations with recovery from local acute pollution, using the example of the *Torrey Canyon* oil spill of 1967 and its subsequent "clean-up" using toxic industrial cleaning agents as dispersants (hereafter we will refer to these cleaning agents as dispersants – at the time these were widely called detergents, e.g. Smith 1968; retrospectively, they are now referred to as Type 1 dispersants). Finally, we discuss how more than two stressors can interact, using the example of acute local pollution from the *Torrey Canyon* incident, chronic regional pollution from Tributyltin-based anti-fouling paints, and climate fluctuations and recent change. We use these case studies to make the case for long-term and broad-scale observations for better understanding of recovery from both acute and chronic pollution. We highlight the importance of considering "baseline" or "reference" conditions as envelopes viewed over appropriate timescales. We also emphasise that existing, known and newer emerging pollutants must be viewed in the context of multiple interacting impacts on the marine environment.

2 Case studies

2.1 Case Study 1 – interactions of overfishing and climate change in the English Channel

Fluctuations in climate have long been known to drive changes in fish populations and assemblage composition (Cushing, 1973), particularly in pelagic species. Since Medieval times, in the English Channel climate fluctuations have led to 'switches' between cold-water Atlantic herring (*Clupea harengus*) and warm-water European pilchards (also known as sardines, *Sardina pilchardus*) (Hawkins et al., 2003, 2013; Russell et al., 1971; Southward, 1980; Southward et al., 1988). The changes before the 19th century occurred before fishing became sufficiently industrialised to influence pelagic fish stocks. For bottom-living species, fishing pressure has been ever-increasing since the advent of trawling in the sailing era, exacerbated by the onset of mechanised industrial-scale fishing in the late 19th Century (Jennings and Kaiser, 1998; Robinson, 1996; Russell, 1942; Thurstan et al., 2014). Signs of declines in fish catches as a consequence of overfishing were evident by the 1880s (Garstang, 1900; Russell, 1942; Sims and Southward, 2006). This led to the first formal fisheries investigations and the formation of research bodies in the UK. These included the Marine Biological Association of the UK (MBA), established in 1884 to pursue the study of marine life, both for its scientific interest and because of the need to know more about the life-histories and habits of food fishes (Southward and Roberts, 1984).

The various investigations by the MBA – commenced in the late 1880s and given new momentum in the 1900s by the International Investigations of the newly-formed International Council for the Exploration of the Seas (ICES) – provide an excellent fisheries-independent database, based on standard hauls around sampling station L4 in the western English Channel off Plymouth (Mieszkowska et al., 2014b; Southward et al., 2004). Records spanning over 100 years have been invaluable for attempting to disentangle the influence of climate fluctuations (Fig. 1a) and more recent rapid climate change from increasing fishing pressure (Genner et al., 2010, 2004; Hawkins et al., 2003, 2013; Southward et al., 2004). The latter has occurred as fishing switched from an artisanal craft, powered by wind and oars, to an industry, powered by coal and later oil. The fishing industry expanded in response to rapid growth of remote urban markets, aided by refrigeration and better transport links, particularly railways.

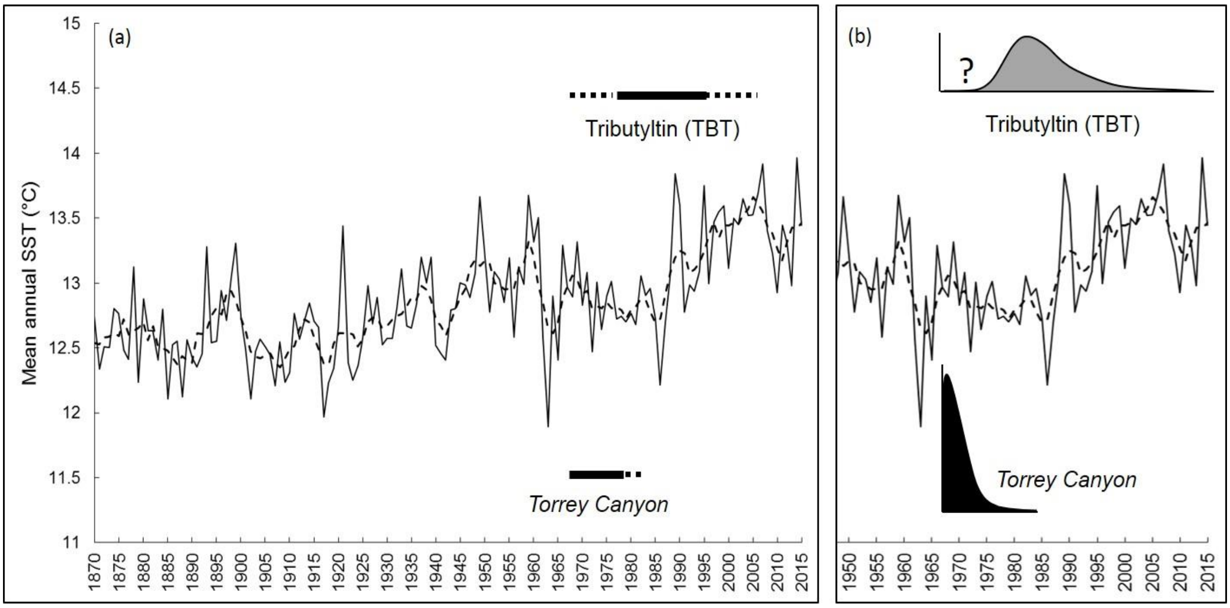

Fig. 1 (a) Mean annual Sea Surface Temperature (SST) 1870–2015 off Plymouth, UK (data from Met Office Hadley Centre) (solid line: annual mean temperature; dashed line: 5-year running average); (b) Sea Surface Temperature during the time window of detailed observations on rocky shores in southwest England from 1950s to date. In (a) duration of acute pollution from the *Torrey Canyon* oil spill and chronic pollution from Tributyltin (TBT) based antifouling paints is indicated by bars (solid bar: greatest impact; dotted bar: onset or recovery). In (b) the onset and severity of acute pollution from the *Torrey Canyon* and chronic TBT pollution in southwest England are represented schematically (y-axis: estimated severity of impact; "?": unknown onset of pollution).

During the 130-year duration of scientific research and monitoring at the MBA, considerable changes have occurred in the composition of bottom-fish assemblages off Plymouth (Genner et al., 2010, 2004; McHugh et al., 2011). The abundance of skates and rays (Rajidae) greatly reduced through the twentieth century (Fig. 2a), largely because their slow growth, late maturation and low fecundity rendered them vulnerable to overfishing (Dulvy and Reynolds, 2002). Some species, such as the common skate (*Raja batis*) and angel shark (*Squatina squatina*) that were once plentiful are now biologically-extinct in the English Channel, as well as the Irish Sea (Brander, 1981; Griffiths et al., 2010; McHugh et al., 2011). Observations off Plymouth are mirrored across European shelf seas

(Dulvy et al., 2000), but the extended duration of the Plymouth datasets allows interpretation of when fisheries-induced biodiversity loss has taken place.

Many of the observed changes in assemblage composition in the Plymouth time-series have been driven by climate. For example, catches of bream (Sparidae) (Fig. 2b) reflect climate fluctuations well: rarely being caught during cold periods (i.e. 1900s–1920s and 1960s–early 1980s), whilst being much more frequently caught during the warmer periods (1950s). Recent catches (2000 onwards) have been much higher than previously recorded, as befits a period when human-driven rapid warming has been recognised – although catches have dropped off a little since the recent colder period 2009–2013 (see Fig. 1). Such changes were reflected in shifts in whole assemblage composition, strongly correlated with temperature (Genner et al., 2010, 2004). The effects of temperature have been most notable in small-bodied species, suggesting that the climate-responses of larger-bodied species have been masked in the region due to the overwhelming pressure of fishing (Genner et al., 2010).

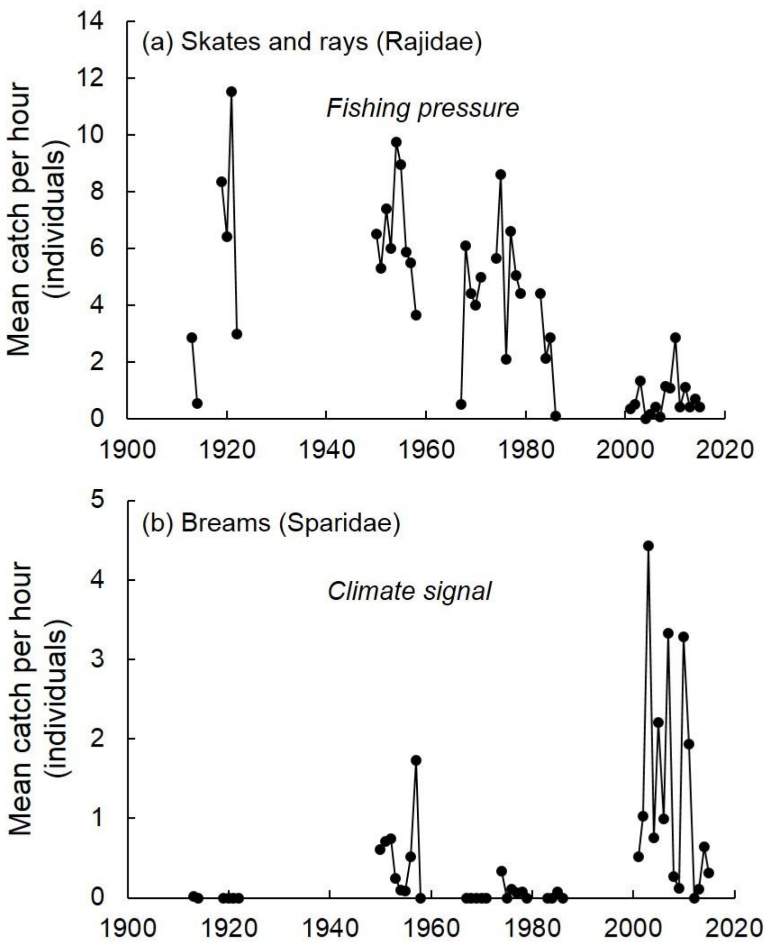

Fig. 2 Mean annual catches of (a) skates and rays (family: Rajidae) showing vulnerability to fishing pressure and (b) breams (family: Sparidae) showing response to climate, recorded over 100 years from MBA standard hauls around the L4 sampling station off Plymouth (see Genner et al., 2004, 2010 for details).

The observed changes off Plymouth mirror those seen more widely across the European continental shelf. Species with typically southern distributions have locally increased in abundance, whilst those with typically northern distributions have locally declined (Simpson et al., 2011). It is possible that climate change and fishing have had compounding interactive effects on stocks, as has been reported in other systems (Lindegren et al., 2013). Several mechanisms have been proposed. For example, rapid climate change can influence phenological cycles, leading to mismatches of larvae with planktonic food and subsequent poor recruitment, thereby reducing resilience to overfishing (Beaugrand et al., 2003; Edwards and Richardson, 2004). Additionally, or alternatively, resilience to climate change may be reduced in overfished stocks that possess lower portfolio of genetic and phenotypic diversity (Planque et al., 2010; Schindler et al., 2010), potentially increasing the probability of mismatches between larval release and larval food supply, that can lead to poor recruitment (Cushing, 1973).

Climate change has led to apparent geographic shifts in the distributions of some fish species on the European continental shelf (Montero-Serra et al., 2015; Perry et al., 2005). Such gains at the poleward range edge are expected to be matched by losses at the southern range edge. However, evidence over recent years is more suggestive of climate-driven changes in abundance of existing species occurring locally and over decadal scales, rather than rapid regional-scale invasions and extirpation of species as a consequence of thermal variability. Such changes may also not be spatially consistent (Simpson et al., 2011). Individual fish do not necessarily track optimal thermal environments for growth (Neat and Righton, 2007). Taken together, this pattern is suggestive of responses to climate change being complex and mediated by the numerous context-dependent variables, including the availability of suitable habitat at appropriate depth (Rutterford et al., 2015), as well as abundance of appropriate food resources, and the impact of antagonists (fisheries, predators, competitors and pathogens).

2.2 Case Study 2 – the Torrey Canyon *oil spill*

The *Torrey Canyon* ran aground off the coast of Cornwall, southwest England, on 17 March 1967, containing nearly 120,000 T of Kuwaiti crude oil. This was the first major oil spill in UK waters and the largest from a tanker to date at that time (see Table 1 summarising selected major oil spill incidents), prompting a major clean-up response. There was much political interest at the highest level with the then Prime Minister, Harold Wilson, owning a holiday home on the nearby Isles of Scilly. The armed forces were mobilised (see Smith, 1968 for overview of events): the Navy coordinated the response; the Air Force bombed the wreck once salvage operations failed; the Navy tracked and sprayed dispersant on the oil at sea; the Army were deployed to clean it up once ashore. At the time, there was no past experience to draw upon; nor was there a plan to hand for dealing with such a spill. The subsequent practice of oil spill contingency planning and specialist response units (e.g. the Oil Pollution Research Unit, Oil Spill Response Limited, the International Tanker Owners Pollution Federation) were a direct result of this and other incidents. Regulations and guidelines have also been developed by the International Maritime Organisation, reinforced by international agreements (e.g. the International Convention for the Prevention of Pollution from Ships (MARPOL) 1973).

An emergency committee and task force under the overall direction of the Navy was established. The Marine Biological Association of the UK (MBA) was also mobilised to provide scientific advice (see Smith 1968 for details). The scientific staff of the MBA were tasked with emergency investigations into the toxicity of the oil, the dispersants used in the clean-up operation, and the oil-dispersant mixture. MBA staff members were also employed to survey the immediate environmental effects at sea and on the shore. *Sarsia*, the research vessel of the MBA, was dispatched and arrived on-site within 10 days of the grounding. The crew recorded the extent of the slick and investigated tainting of bottom fish by oil under the slick and also the impacts on birds (see Smith, 1968 for full account).

Table 1 Selected oil spill incidents from ships and installations (various sources including International Tanker Owners Pollution Federation (ITOPF) database).

Incident	Date	Location	Oil Spilt (Tonnes)
Torrey Canyon	1967	West Cornwall, UK	118,000
Amoco Cadiz	1978	West Brittany, France	223,000
Exxon Valdez	1989	Alaska, USA	38,000
Arabian Gulf installations	1991	Kuwait	> 1 million
Braer	1993	Shetland Islands, UK	85,000
Sea Empress	1996	Pembrokeshire, UK	72,000
Erika	1999	Bay of Biscay, France	20,000[a]
Prestige	2002	Cape Finisterre, Spain	63,000[a]
Hebei Spirit	2007	Taean, South Korea	11,000
Deep Water Horizon installation	2010	Gulf of Mexico, USA	~ 7 million

[a]Heavy fuel oil

On accessible shorelines, excessive amounts of toxic organic solvents were sprayed onto the oil when it reached the shore, killing virtually all intertidal grazers, particularly limpets of the genus *Patella* (Hawkins and Southward, 1992; Smith, 1968; Southward and Southward, 1978). Along less accessible stretches of coast, drums of dispersant were pitched over the clifftops to rupture on the rocks below. In total around 10,000 T of dispersant were applied to the 14,000 T of oil estimated to have come ashore in west Cornwall (Smith, 1968). The crude oil proved considerably less toxic to marine life than the dispersants applied and the oil-dispersant mixture (Smith, 1968). The subsequent recovery of shores from the spill (Southward and Southward, 1978) mirrored the succession demonstrated following classic experimental removals of limpets (Burrows and Lodge, 1950; Jones, 1946, 1948; Lodge, 1948; Southward, 1964; see Hawkins and Hartnoll, 1983a for review). On shores where dispersant was applied, ephemeral, predominantly green species of algae (*Ulva* spp.) appeared, followed approximately a year later by a massive recruitment of fucoid algae (mainly *Fucus vesiculosus*, but also *F. spiralis* and *F. serratus*; Fig. 3a). The fucoid canopy had a negative effect on the barnacles that had survived by facilitating large numbers of dogwhelks (*Nucella lapillus*) that fed on the barnacles (Hawkins and Hartnoll, 1983b). The canopy also prevented further barnacle recruitment by fucoid sweeping (Hawkins, 1983; Jenkins et al., 1999). The abundant growth of *Fucus* provided an excellent nursery for recruitment of limpets (*Patella vulgata*), which settled and prospered under the canopy (Southward and Southward, 1978; Fig. 3b). Over time the larger plants died, dislodged by wave action (Jonsson et al., 2006) or eaten by the limpets (Notman et al., 2016). This led to a period in which the abundance of canopy algae massively declined (Fig. 3a) due to the large population of limpets preventing fucoid recruitment (Hawkins and Southward, 1992; Southward and Southward, 1978). The normal occurrence of occasional small natural escapes of fucoids in patches of low limpet density or amongst dense barnacles or mussels (Crowe et al., 2011; Hawkins, 1981; O'Connor and Crowe, 2008) did not occur due to intense grazing pressure. At Porthleven, south Cornwall, starving limpets abandoned their normal homing habits and migrated across the shore in a large front in search of food, before dying (Hawkins and

Southward, 1992; Southward, 1979; Southward and Southward, 1978). Stabilisation of the shore community occurred via an aphasic damped oscillation (Hawkins et al., 1983; Hawkins and Southward, 1992; Southward and Southward, 1978) between the dominant space occupying seaweeds (*Fucus* spp.) and the main grazers (*Patella* spp., particularly *P. vulgata*) (Fig. 3).

Southward and Southward (1978) estimated that most shores took up to 10 years to return to what they considered a pre-spill state (Table 2, modified from Hawkins and Southward, 1992). Godrevy (a site managed by the National Trust, a major land-owning conservation charity in the UK), although affected by oil, was not sprayed with dispersants because of concerns about the resident colony of grey seals (*Halichoerus grypus*). Here, recovery appeared to occur much more quickly, within 2–3 years (Table 2; Hawkins and Southward, 1992; Southward and Southward, 1978). At Porthleven, one of the most heavily-sprayed shores, a longer study suggested that it took between 13 and 15 years to return to normal (Table 2; Hawkins and Southward, 1992; Hawkins et al., 1983).

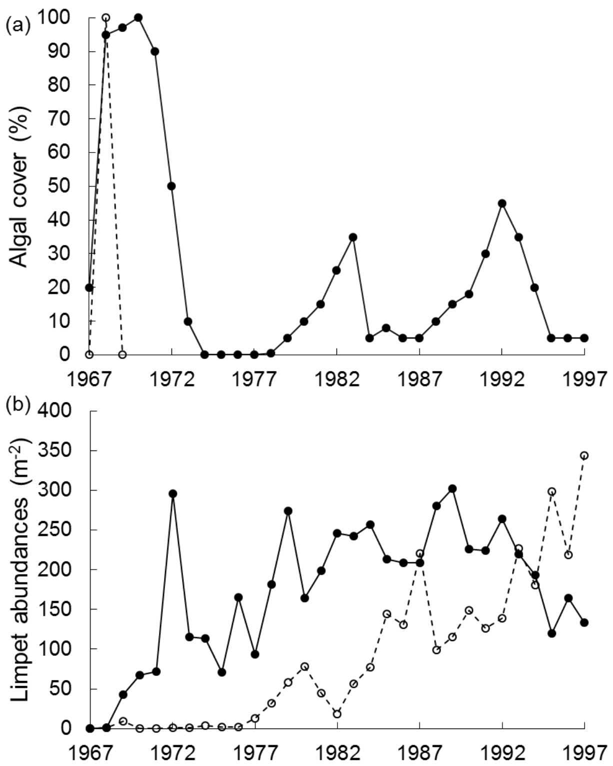

Fig. 3 Recovery of rocky shore community at Porthleven following the *Torrey Canyon* oil spill incident, illustrated by fluctuating abundances of: (a) green ephemeral (dotted line) and fucoid canopy (solid line) algae; and (b) warm-water limpets (*Patella depressa*) (dotted line) and cold-water

limpets (*Patella vulgata*) (solid line). *(Adapted from Hawkins and Southward, 1992; Southward and Southward, 1978; updated with new data.)*

Table 2 Timescale in years of recolonization of rocky shores in Cornwall following the *Torrey Canyon* incident. "*" indicates relatively High (***), Moderate (**) and Low (*); "n/a" indicates not applicable. *(Modified from Hawkins and Southward, 1992; Southward and Southward, 1978.)*

	Lizard Point Exposed	Lizard Point Sheltered	Porthleven	Maen Du Point	Sennen Cove Exposed	Sennen Cove	Cape Cornwall	Godrevy	Trevone Exposed	Trevone Sheltered
Relative exposure to wave action	***	**	**	**	***	**	***	**	***	*
Amount of oil stranded	*	**	***	*	**	***	**	**	**	**
Dispersant treatment	*	***	***	**	**	***	**	0	**	***
Persistence oil/oil-dispersant mix	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Max. ephemeral green algae	1	1	1	1	0-1	1	1	None	0-1	1
Max. *Fucus*	2-3	1-3	1-3	2-3	1-3	1-3	1-2	None	2	1-3
Min. barnacles	2	2	3	4	3	3	3	< 6 mo.	2	2-6
Max. *Patella*	?	6	5	5	?	3	3	n/a	3	5
Fucus starts of decline	4	4	4	4	4	5	3	n/a	3	4
Fucus all gone	5	6-7	6-7	6	5	6	6	n/a	5	8
Barnacles increased	4	6	6	5	4	6	4	1	3	7
Patella reduced	?	6	8	7	6-7	8	7	n/a	6	n/a
Time for recovery (normal species richness regained)	5	9	> 10 (probably 13-15)	8-10	9	9	8-9	2	5-6	> 9-10

Broader-scale contextual monitoring and more recent experimentation and modelling can also help explain some of the patterns observed during the recovery period, particularly with regard to the balance of barnacle and limpet species. Following the extremely cold winter of 1962–63 (compared with the warmer 1950s), the Northeast Atlantic entered a colder period until the mid-1980s (Fig. 1). During this period, the cold-water barnacle, *Semibalanus balanoides*, became more abundant and warm-water barnacles, *Chthamalus* spp., became scarcer (Mieszkowska et al., 2006, 2014a; Poloczanska et al., 2008; Southward, 1967, 1991; Southward et al., 1995). The warm-water limpet, *Patella depressa*, was also much slower to recover from the oil spill incident than the colder-water limpet, *P. vulgata* (Fig. 3b). This was probably due in part to the direct effects of a colder climate

(Hawkins et al., 2008; Kendall et al., 2004; Southward et al., 1995) leading to lower abundance throughout its northern range in the British Isles in general and west Cornwall in particular. Subsequent experimental work (Moore et al., 2007) has also shown that unlike *P. vulgata*, *P. depressa* does not prosper under macroalgal canopies. The dense cover of *Fucus* following the oil spill probably facilitated recovery of *P. vulgata* at the expense of *P. depressa*. Only since 1990 has *P. depressa* become common again at Porthleven, along with the other shores in the southwest of England, especially in years when fucoid cover was low (Hawkins et al., 2008; Hawkins unpublished observations; Fig. 3).

2.3 Case Study 3 – interaction between acute oil spill pollution, climate change and chronic Tributyltin pollution

Another species negatively affected by the *Torrey Canyon* incident was the hermit crab, *Clibanarius erythropus*. This species was first recorded in the southwest of England at the very end of the warm period that culminated in 1959 (Carlisle and Tregenza, 1961), representing an extension of its northern range edge across the English Channel from Brittany, France. Following the acute impact of the oil spill and subsequent application of toxic dispersants, the population was severely affected at several badly polluted sites in west Cornwall, including Marazion in Mount's Bay (Southward and Southward, 1977, 1988; Fig. 4a). Declines were also recorded, however, at an unaffected reference site approximately 100 km to the east at Wembury, near Plymouth (Southward and Southward, 1977, 1988; Fig. 4b). Eventually *C. erythropus* disappeared completely from the British fauna by the mid 1980s (Southward and Southward, 1988). This was attributed to lack of recruitment in a northern range-edge population due to the colder conditions in the 1960s, 1970s and early 1980s. Southward and Southward (1988) also suggested that this decline may have been exacerbated by reductions in local populations of the dogwhelk, *Nucella lapillus*, and hence supply of its preferred home shell, due to chronic Tributyltin (TBT) pollution. TBT pollution led to masculinisation of female dogwhelks ("imposex"), in severe cases leading to sterility and death as a proliferating *vas deferens* blocked the female genital duct (Bryan et al., 1986; Gibbs and Bryan, 1986). Populations of *N. lapillus* thus became much reduced in the southwest of England during the 1970s and 80s (Bryan et al., 1986; Spence et al., 1990).

The lack of empty available dogwhelk shells may have delayed the recovery of *C. erythropus* hermit crabs from declines triggered by the *Torrey Canyon* incident and a cold period, despite conditions becoming warmer from 1989 onwards. The use of TBT-based anti-fouling paints has been progressively banned since 1985 in the UK and France, and subsequently elsewhere (Sonak et al., 2009). This eventually led to an International Maritime Organisation (IMO) global ban in 2008, although sediment hotspots persist (Eggleton and Thomas, 2004) and continued illegal sale and use has occurred (Turner and Glegg, 2014). Since 1985, recovery of dogwhelk populations has been slow but steady (Fig. 5). Despite some legacy pollution issues due to contaminated sediments, populations in the southwest of England had largely recovered to low levels of imposex a decade later, by around the year 2000. A re-survey in 2016 confirmed this, with imposex being undetectable in the previously routinely monitored populations (see also Langston et al., 2015; Matthiessen, 2013; Nicolaus and Barry, 2015; Wilson et al., 2015).

Interestingly, *C. erythropus* were found again on rocky shores at Marazion in spring and summer 2016 (Fig. 4a), suggesting a recruitment event had at last occurred, possibly in 2015, from populations in northern France. They also returned to Wembury in 2016 but are currently very rare (Fig. 4b). In parallel, there has been some recruitment in 2015 in west Cornwall of the stalked barnacle, *Pollicipes pollicipes*, another species whose nearest breeding populations are in Brittany (Crisp and Fischer-Piette, 1959; Hawkins, Mieszkowska and Hiscock, unpublished observations). Thus *C. erythropus* is an example of a species that was heavily impacted by an oil spill incident at a local spatial scale; whose recovery was delayed at its range edge by poor recruitment during a period of colder weather; perhaps compounded by lack of home shells due to regional-scale chronic pollution impacts on *N. lapillus*.

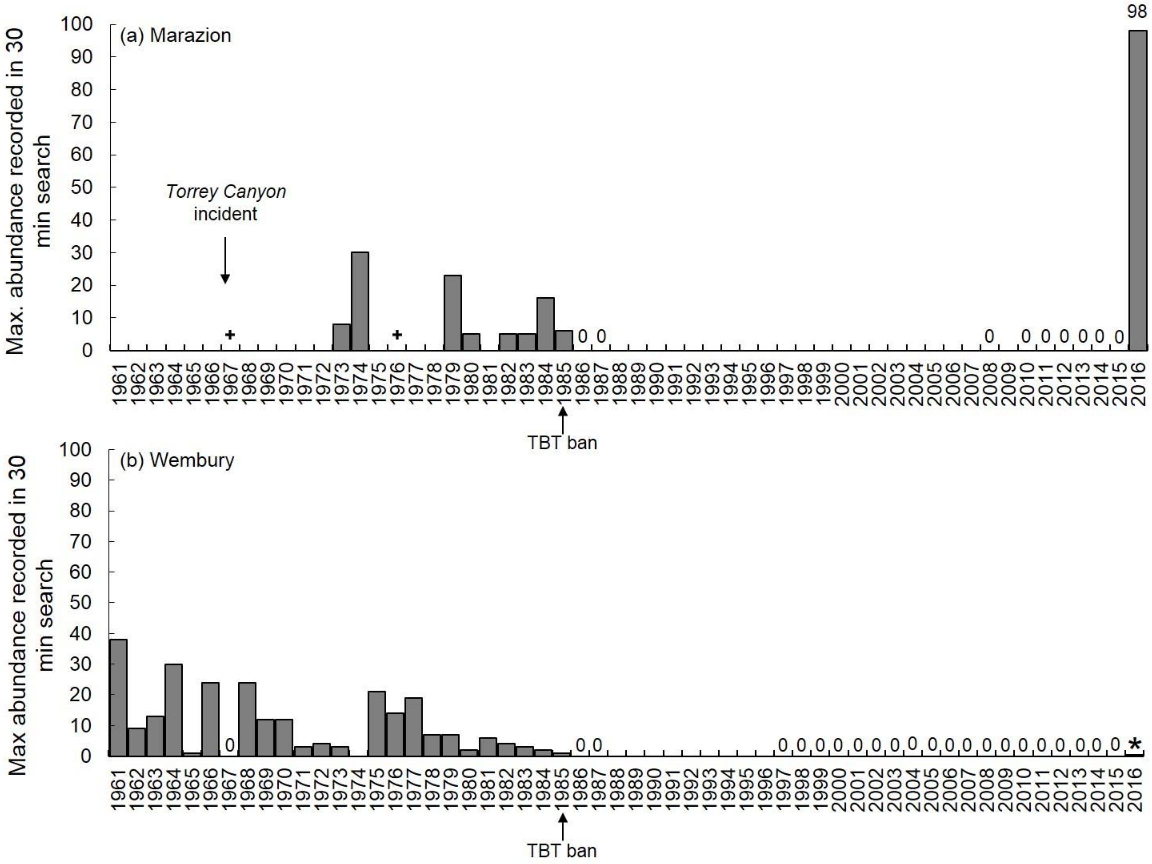

Fig. 4 Maximum abundance found in any year of *Clibanarius erythropus* recorded in 30 min searches at (a) Marazion, Cornwall (affected by *Torrey Canyon* incident), and (b) Wembury, Devon (unaffected by *Torrey Canyon* incident), between 1961 and 2016. "0": zero count as opposed to no record; "+": present but not quantified; "*": single individual found in 1-hour search at Wembury in September 2016. *(Modified from Southward and Southward, 1977, 1988 with additional observations.)*

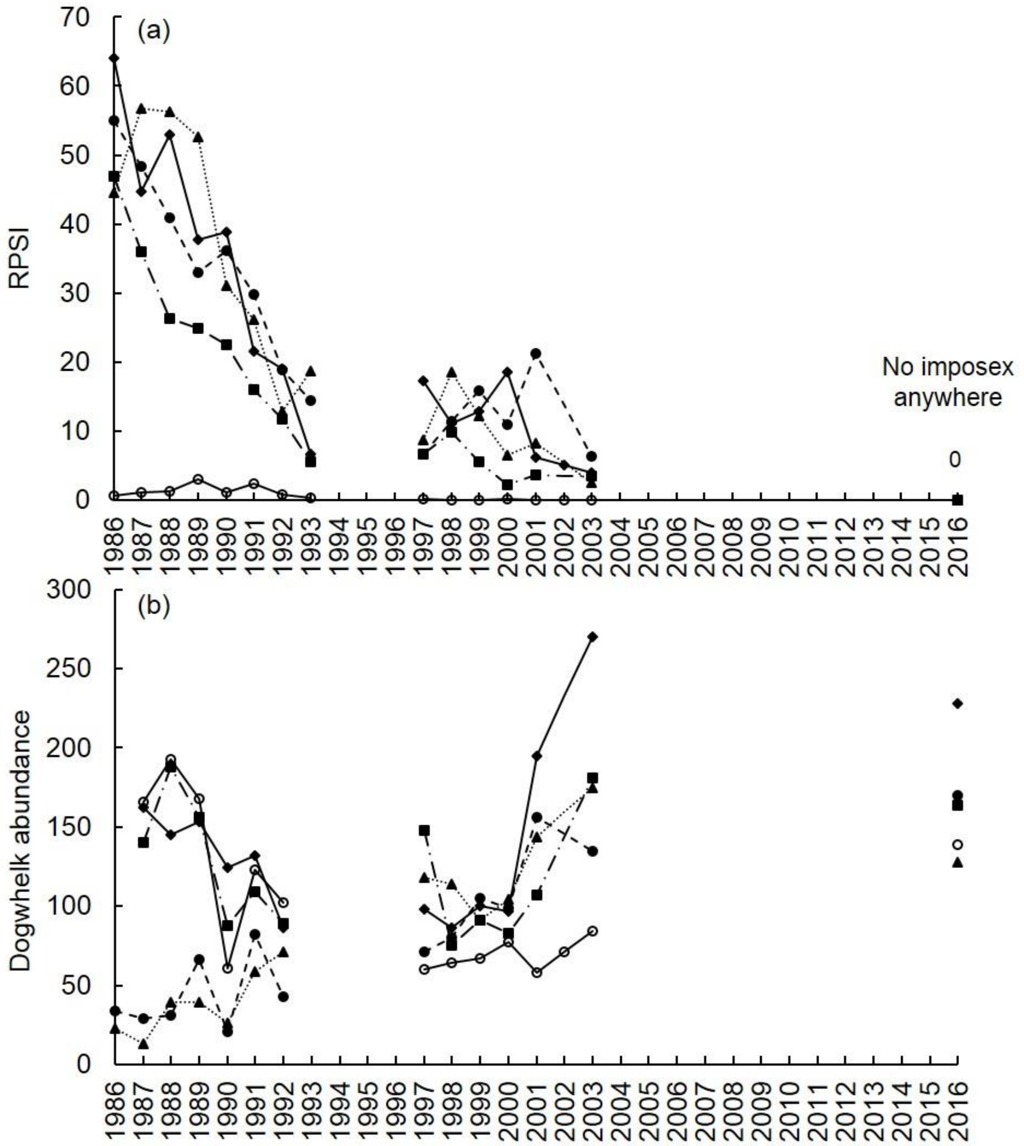

Fig. 5 Recovery of dogwhelk, *Nucella lapillus*, populations around the southwest of England following chronic Tributyltin (TBT) pollution and its subsequent ban, illustrated by: (a) Relative Penis Size Index (RPSI) – an index of imposex in female dogwhelks; and (b) abundance counts. Data recorded from five routinely monitored sites: Tregantle (◆), St Agnes (○), Renney Rocks (■), Jennycliff (▲) and Kingsand (●). St Agnes can be viewed as a control site as it had very low incidence of imposex at the time of peak contamination. *(Modified and extended from Bray, 2005; Hawkins et al., 2002.)*

3 Discussion

3.1 Role of long-term observation in understanding impacts and recovery

Following an acute pollution incident or cessation of a chronic impact, recovery in marine ecosystems largely occurs by colonisation from remote sources, in many species via pelagic larvae or widely-dispersed propagules of algae. Such recovery occurs against a backdrop of considerable natural variability driven by seasonal cycles and climate fluctuations, upon which rapid anthropogenic climate change has been superimposed in recent decades. Long-term data from impacted and, where they exist, control or reference sites are essential for interpreting trajectories and timescales of recovery (Hawkins et al., 2013; Mieszkowska et al., 2014b). Such data have been available for rocky shores in southwest Britain since the 1950s (Mieszkowska et al., 2006, 2014a; Poloczanska et al., 2008; Southward, 1991, 1967, Southward and Crisp, 1954; Southward et al., 1995, 2004;). These studies have proven to be invaluable resources for increasing the understanding of the ecological processes involved in recovery from the *Torrey Canyon* oil spill in 1967 and the subsequent clean-up using highly toxic first generation dispersants (Hawkins et al., 2002, 1983; Hawkins and Southward, 1992; Smith, 1968; Southward and Southward, 1978). Such observations are also essential for better understanding of recovery from chronic pollution (Hawkins et al., 2002) as well as to disentangle the effects of overfishing from the influence of climatic fluctuations and recent rapid climate change (e.g. Genner et al., 2010).

In parallel with the long-term observing there has also been a large body of experimental work unravelling the interactions of the main species on British and European rocky shores, especially the role of patellid limpets as keystone grazers in structuring biological communities (Coleman et al., 2006; Hawkins, 1981; Hawkins and Hartnoll, 1983a; Jenkins et al., 2005; Jones, 1948; O'Connor and Crowe, 2005; Southward, 1964). How these interactions are, in turn, modulated by climate change, or conversely ameliorate the direct effects of climate change, have also been elucidated (Moore et al., 2007), leading to qualitative (Hawkins et al., 2009, 2008) and quantitative (Poloczanska et al., 2008; Spencer et al., 2012) models of the predicted influence of climate change on species interactions and implications for ecosystem functioning. These insights have proven invaluable in understanding the processes involved in the recovery of rocky shores from the *Torrey Canyon* oil spill and chronic TBT pollution.

3.2 Baselines or envelopes?

The examples presented in this paper illustrate the importance of sustained ecological observations, which are also referred to as monitoring or surveillance. It is important to distinguish sustained observing from compliance monitoring that is driven by statutory powers or legislation. Compliance monitoring often works by measuring the frequency with which a standard is exceeded. Such data, whilst useful, can be insufficient when trying to measure impacts on community structure or ecosystem functioning – especially as both environmental standards and analytical instrumentation tend to change over time. Langston et al. (2006) discussed the problems of using compliance monitoring for judging the health of the ecosystems in candidate areas for conservation with reference to EU designated Special Areas of Conservation.

The examples chosen also emphasize that there is no such thing as a "baseline" condition, as conditions fluctuate over time due to natural processes, upon which anthropogenically-driven climate change is now superimposed. A more realistic view of a reference state may be a multi-dimensional "envelope" of conditions that captures normal spatial and temporal fluctuations. Only with long-term data can such envelopes be defined. This is essential when placing marine pollution in the context of other pressures in coastal and oceanic systems and managing multiple stressors.

3.3 Pollution in the context of other impacts

It is possible to broadly place various types of pollution into the context of other global, regional and local impacts on the marine environment (Table 3). We have endeavoured to summarise the scale at which an impact occurs and its degree of severity. Furthermore, global change interacts with and exacerbates regional- and local-scale impacts (Hawkins, 2012). Fishing pressure clearly interacts with climate change. Harmful algal blooms are probably increasing due to regional-scale eutrophication and greater stratification of coastal waters due to warming (Hallegraeff, 2010; Paerl and Huisman, 2008). There is evidence that jellyfish blooms are increasing due to multiple causes, including climate change (Purcell, 2012), fishing down the food web (Daskalov et al., 2007; Jackson et al., 2001) and proliferation of artificial structures (Duarte et al., 2013; Makabe et al., 2014).

National and regional governments, in concert with international bodies, have a generally good record in dealing with pollution issues once identified, properly diagnosed, acted upon and subsequently monitored. The scientific community also has an excellent record of prompting governments to act. Success stories include identification of the pernicious influence of some pesticides (Carson, 1962), dealing with sewage pollution in fresh and coastal waters (EU Water Framework Directive), and identifying and dealing with TBT pollution (Langston et al., 2015; Nicolaus and Barry, 2015). Emerging pollution issues are sometimes identified by scientists and then taken up by campaigning organisations in civil society. A good example of this has been pollution by plastics, including micro-plastics, which are in the course of being banned from certain products by the UK government at the time of writing in 2016. In the UK, niche stakeholders such as Surfers Against Sewage (https://www.sas.org.uk/) have been particularly active, including campaigning against long and short sea outfalls, and more recently against plastic pollution, building on earlier work by the popular beach cleaning initiatives of the Marine Conservation Society (https://www.mcsuk.org/).

Pollution is in some ways easier to deal with than some other anthropogenically-driven environmental pressures (see Table 3). A compound can be chemically characterised and concentrations measured in the laboratory and in the field; toxicity can be assessed relative to permissible standards defined at either discharge point or background levels. Clearly this becomes much more difficult with complex mixtures from multiple and often diffuse sources, but if society and governments are willing, problems can be identified and solved.

Other impacts are much more difficult to manage, especially if operating on a global scale. This is often because there is little feedback between individual and local actions and mitigation of global impacts. Many, such as overfishing – whether global, regional or local – are tragedies of commons. Both global environmental change and overfishing are good examples. Pressures of economic development, coupled with food security for a better quality of life, together drive massive environmental degradation on extremely long and difficult-to-reverse timescales. The very slow but recent recovery of the "northern" Atlantic cod populations off Newfoundland and Labrador is a testimony to the hysteresis associated with the overexploitation of species (Rose and Rowe, 2015). More intangible and long-term impacts such as ocean acidification have rapidly caught the attention of the scientific community and governments (Galaz et al., 2012), but are so long-term that they are yet to strongly register with society, despite the high certainty of considerable impact.

Piecemeal modification of habitat driven by a multiplicity of developments and extractive activities in the coastal zone and offshore combine to drive regional-level change (Airoldi and Beck, 2007; Dafforn et al., 2015; Firth et al., 2016). These local impacts range from land claim, defence of existing land reclamation, new transport infrastructure for shipping, road and rail, and prime waterside real estate. Local interventions, often decided at the individual property-owner or municipality level, can rapidly scale-up to the regional scale (Airoldi et al., 2005). Such interventions have been shown to interact with other pressures, such as the spread of non-native species (Airoldi et al., 2015; Bulleri and Airoldi, 2005; Firth et al., 2011; Mineur et al., 2012). Sea defences have also been built as an adaptational response to rising and stormier seas, demonstrating a local impact prompted by a response to global change.

To increase ecosystem resilience, conserve natural capital and maintain valuable goods and services it is important to focus action on what is manageable in the short- (1–5 years) and medium- (5–25 years) term, whilst continuing to be vigilant by monitoring long-term changes. Should humanity massively reduce its reliance on fossil fuels and stabilise greenhouse gas emissions, the inertia in the global climate system (Solomon et al., 2009) is such that it is likely to take much more than 50 years for temperature and ocean pH to stabilise within reasonably safe boundaries. Therefore, reducing impacts that are easier to manage, many by initiating local or regional actions, will convey resilience to global change. Biosecurity can be heightened to combat invasive species; overfishing can be reduced; planning systems can be used to combat habitat loss; and restoration and rehabilitation approaches can be applied to improve local ecosystems. The extent to which many sources and impacts of pollution have been dealt with successfully demonstrates a good track record in adaptive management in response to anthropogenic stressors. Our view is that overfishing, the global pollution of greenhouse gases leading to climate change and ocean acidification, invasions of non-native species and habitat loss can all be considered more worrying threats than many types of regional- and local-scale pollution. But only by managing those impacts over which we have control at the national and regional level – such as pollution and habitat degradation – can we confer resilience to global change over the next 40–50 years. This is essential whilst decarbonisation of human society is underway, to keep our planet within the safe limits signed up to by the international community at the Paris summit of 2015.

Table 3 Some examples of the scale of occurrence of marine and coastal environmental issues, their severity ("✶✶✶": high; "✶✶": moderate; "✶": low) and trends ("↑": worsening; "↓": ameliorating; "↑↓": variable or different types of impacts within category showing opposite trends; "?": unknown)

Environmental issue	Scale: Global (10^3 km)	Scale: Regional (10^2 km)	Scale: Local (<10 km)	Cause/Source	Comments	Selected references
Global change						
Anthropogenic climate change: warming, rising and stormier oceans	✶✶✶ ↑	✶✶✶ ↑		Population growth leading to industrialization, agricultural intensification and greenhouse gas emissions.	Some regions are particularly vulnerable as demonstrated by velocity of climate change studies.	Burrows et al., 2011; Cheung et al., 2009; Edwards and Richardson, 2004; Harley et al., 2006; Helmuth et al., 2014, 2006; IPCC, 2014; Loarie et al., 2009; Rahmstorf et al., 2015; Walther, 2010
Ocean acidification	✶✶ ↑	✶✶✶ ↑		CO_2 emissions.	A global phenomenon with some regions particularly vulnerable such as upwelling areas.	Connell et al., 2013; Doney et al., 2009; Feely et al., 2008; Gaylord et al., 2015; Hall-Spencer et al., 2008; Harvey et al., 2013; Kroeker et al., 2010; Orr et al., 2005; Raven et al., 2005; Sabine et al., 2004; Wootton et al., 2008
Non-native invasive species	✶✶ ↑	✶✶ ↑	✶ ✶↑	Maritime trade, aquaculture, deliberate introductions.	A global phenomenon with hot spots in ports and marinas.	Airoldi et al., 2015; Bax et al., 2003; Hulme, 2009; Mineur et al., 2012; Rius et al., 2014; Ruiz and Carlton, 2003; Stachowicz et al., 2002a, 2002b; Streftaris et al., 2005
Overharvesting of living resources						
Over-exploitation of large apex species:						
- Pelagic predators	✶✶ ↑	✶✶✶ ↑		Overfishing driven by high market	A worldwide problem with some regions particularly	Baum et al., 2003; Jackson et al., 2001; Myers and Worm, 2003;

				prices.	vulnerable.	Scheffer et al., 2005; Worm and Tittensor, 2011
- Whales	✶✶✶ ↓	✶✶ ↓	✶ ↓		Protection has led to recovery but some populations still low compared to pre-whaling.	Alter et al., 2012; Constantine et al., 2012; Monsarrat et al., 2016
Overfishing	✶✶ ↑	✶✶✶ ↑↓	✶✶✶↑	Overfishing driven by population growth & high market prices.	Shelf seas overfished globally. Regional-scale impact scaling up worldwide, especially bottom fish. Evidence of regional stock recovery in places.	Daskalov 2002; Daskalov et al., 2007; Jackson et al., 2001; Pauly et al., 2002; Rose and Rowe, 2015; Thurstan et al., 2010; Worm et al., 2009
Examples of Pollution						
Microplastics	✶✶ ✶↑?	✶✶✶ ↑	✶✶✶ ↑	Use in products, breakdown of plastic litter.	An emerging issue likely to be global in extent but with local hotspots.	Andrady, 2011; Browne et al., 2011, 2007; do Sul and Costa, 2014; Sutherland et al., 2010; Thompson et al., 2004; Yang et al., 2015
Eutrophication		✶✶ ↑	✶✶✶ ↑	Multiple sources including run-off from agriculture, atmospheric deposition and sewage .	Scaling up from local to regional. Occurs worldwide in coastal and shelf waters. May lead to hypoxia and fish kills, resulting in "Dead Zones". According to recent estimations, there are currently over 600 dead zones around the globe.	Bonsdorff et al., 1997; Cloern, 2001; Diaz et al., 2013; Diaz and Rosenberg, 2008; Mayer-Pinto et al., 2015; Nixon, 1995; Smith et al., 1999; Smith and Schindler, 2009
Organotins		✶ ↓	✶✶ ↑↓	Antifouling paints with legacy pollution in sediments, biocides and plasticizers.	Legislation has largely tackled this problem other than legacy pollution from sediments in Europe and North America, but triphenyltin pollution persists along the coasts of Asia.	Hawkins et al., 2002; Ho et al., 2016; Ho and Leung, 2014; Langston et al., 2015; Matthiessen, 2013; Singh and Turner, 2009; Sonak et al., 2009
Nano-particles	???	✶	✶✶↑	Present in a wide	A new and emerging issue –	Andrady, 2011; Canesi et al., 2014;

				range of commercial products (e.g. sunscreens), possibly released into the marine environment during production process, application and via sewage.	likely to be global in extent. However, assessment of the fate and environmental risks of nano-particles has been hindered by difficulties in their detection, characterisation and quantification. Given the increasing trend of production volumes, contamination in coastal environments is expected to increase.	Matranga and Corsi, 2012; Navarro et al., 2008; Nowack and Bucheli, 2007; Zhu et al., 2011
Persistent organics	✶ ↑↓	✶✶ ↑↓	✶✶✶ ↑↓	Multiple sources including biocides, plastics, plasticizers, surfactants, flame retardants, paints, disinfection agents, solvents, etc.	Main types of persistent organic pollutants (POPs) have evolved over time, from chlorinated compounds to brominated and fluorinated ones. Many can bioaccumulate and biomagnify along marine food chains, posing ecological risks to organisms at high trophic levels. Some can undergo long-range transportation via atmosphere and oceans.	Ahrens et al., 2010; Braune et al., 2005; Connell et al., 1998; Houde et al., 2011; Loganathan and Lam, 2011, Mormede and Davies, 2003
Pharmaceuticals and Personal Care Products (PPCPs); Endocrine Disrupting Chemicals		✶✶ ↑	✶✶✶ ↑	With improvement of health care and living standards, there are increases in consumption of PPCPs worldwide. Many of these chemicals cannot be removed by sewage	Although concentrations of PPCPs remain relatively low in marine environments, many of them (e.g. nonylphenol, triclosan) are endocrine disruptors and may pose risks to the sex ratio, reproduction and population fitness of marine life. The release of	Boxall et al., 2012; Gardner et al., 2013; Gaw et al., 2014; Gómez-Canela et al., 2012; McEneff et al., 2014; Rúa-Gómez and Püttmann, 2012

				treatment and are eventually released into the marine environment.	antibiotic residues trigger a global concern on development of antibiotic resistance in microbial communities. The impacts of other commonly used drugs on marine organisms remain largely unknown.	
Persistent inorganics (metals, etc.)	✶	✶✶ ↓	✶✶✶ ↓	Multiple sources including industrial effluents, mine waste, antifouling paints, etc.	Pollution from metals largely under control – mostly local but can scale up in enclosed waters. Mercury a global problem.	Ashton et al., 2010; Bryan, 1971; Jæger et al., 2009; Phillips, 1977; Scheuhammer, 1987; Singh and Turner, 2009; Turner et al., 2001
Sewage		✶ ↓	✶✶ ↓	Untreated and partially treated wastewater discharges increase with population growth.	Decreasing as treatment schemes implemented worldwide. Particular progress in more developed countries, but the pace of improvement in some developing regions (e.g. India, Africa) remain slow.	Cabral-Oliveira et al., 2014; Corcoran et al., 2010; Costanzo et al., 2001; Costello and Read, 1994; Hawkins et al., 1999; Islam and Tanaka, 2004; Kennish, 2002; Littler and Murray, 1975
Oil spills		✶ ↑	✶✶ ↓	Accidents on rigs and ships.	Tanker-based spills becoming less common. Major blowouts (e.g. Deepwater Horizon) can have regional-scale impacts.	Boehm et al., 2014; Burgherr, 2007; Fisher et al., 2014; Gong et al., 2014; Kingston, 2002; Moore, 2006; Suchanek, 1993
Habitat loss and degradation						
Collateral damage from fishing		✶✶ ↑	✶✶✶ ↑	Towed bottom fishing gear damages benthos.	Local hot spots – occurs on regional scale on continental shelf worldwide.	Althaus et al., 2009; Clark et al., 2016; Collie et al., 2000; Hinz et al., 2009; Jennings and Kaiser, 1998; Kaiser et al., 2000; Sheehan et al., 2013
Coastal development		✶✶ ↑	✶✶✶ ↑	Growing population using coastal zone.	Local schemes can scale up to regional level impacts on some coastlines. Worldwide	Airoldi and Beck, 2007; Dafforn et al., 2015; Firth et al., 2016; Hinrichsen, 1999; Neumann et al.,

					problem.	2015; Small and Nicholls, 2003
Coastal defences		✶ ↑	✶ ✶ ✶ ↑	Adaptational response to rising and stormier seas.	Local schemes can scale up to regional level (e.g. Adriatic, Japan).	Airoldi et al., 2005; Airoldi and Beck, 2007; Chapman and Bulleri, 2003; Davis et al., 2002; Dong et al., 2016; Firth et al., 2016, 2013; Koike, 1996
Aquaculture		✶ ↑	✶ ✶	Overfishing leading to switch to aquaculture. Coastal habitats such as lagoons and mangroves being developed for fish farming. Cage culture can impact enclosed waters if carrying capacity exceeded.	A growing problem as capture fisheries level out. Largely local but can scale up in areas with extensive aquaculture industries such as prawn farming or shellfish culture.	Farmaki et al., 2014; Hargrave et al., 1997; Lin and Fong, 2008; Naylor et al., 2000; Páez-Osuna, 2001; Primavera, 2006; Read and Fernandes, 2003

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