Australia
State of the Environment 2011

Factors potentially affecting the resilience of temperate marine populations

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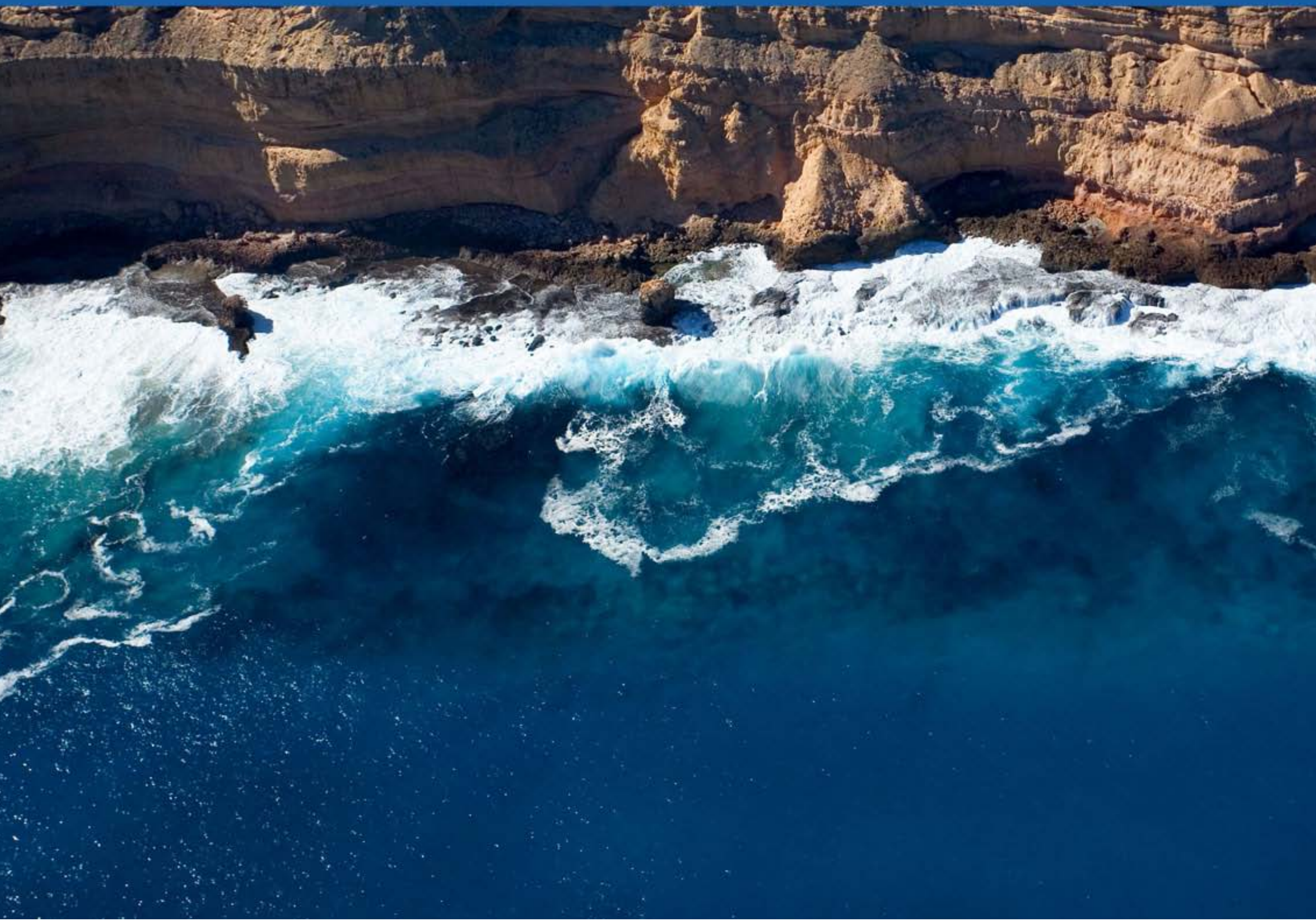
ON BEHALF OF the State of the Environment 2011 Committee



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Cover image

Cliffs at Shark Bay, WA
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Preface

This report was commissioned by the Department of Sustainability, Environment, Water, Population and Communities to help inform the Australia State of the Environment (SoE) 2011 report. As part of ensuring its scientific credibility, this report has been independently peer reviewed.

The Minister for Environment is required, under the *Environment Protection and Biodiversity Conservation Act 1999*, to table a report in Parliament every five years on the State of the Environment.

The Australia State of the Environment (SoE) 2011 report is a substantive, hardcopy report compiled by an independent committee appointed by the Minister for Environment. The report is an assessment of the current condition of the Australian environment, the pressures on it and the drivers of those pressures. It details management initiatives in place to address environmental concerns and the effectiveness of those initiatives.

The main purpose of SoE 2011 is to provide relevant and useful information on environmental issues to the public and decision-makers, in order to raise awareness and support more informed environmental management decisions that lead to more sustainable use and effective conservation of environmental assets.

The 2011 SoE report, commissioned technical reports and other supplementary products are available online at www.environment.gov.au/soe.

Executive Summary

The object of this contribution to the SoE2011 report, is to outline a variety of social and ecological factors that can potentially affect the resilience of temperate marine populations. The practical concepts and issues behind resilience have been illustrated by the way that commercial fisheries for four different Australian fish stocks, western rock lobster, southern rock lobster, eastern gemfish and tiger flathead, have responded to exploitation and other stressors over time.

Six main categories of factors are identified as having the potential to influence population resilience of fish species; these are climate change, fishing, coastal development, introduced marine pests, infectious diseases and the socio-economic influences associated with fishery management.

Climate change is a broad heading for changes that are predicted to occur to our oceanic and meteorological environment in the future. It is predicted that water temperatures around Australia will rise by 1-2°C by 2030 and by 2-3°C by 2070, with the greatest levels of warming in the marine environment occurring in the south east of the country. The East Australian Current (EAC) is expected to strengthen and extend further south than is presently the case and this will intensify the increase in water temperature over its area of influence. Zonal winds are predicted to shift southward and that this will lead to an overall weakening of winds over the southern part of the country, which in turn can be expected to result in a reduction in productivity. The increase in CO_2 levels as a result of fossil fuel emissions will lead to an increase in ocean acidity. Overall, predictions are that climatic conditions will be moving towards an 'El Niño like state' in the future, but there remains uncertainty around the frequency or amplitude of these predicted events.

The warmer water temperatures and overall southward shift in zonal wind and EAC are expected to result in a corresponding southward movement to the distribution of the southern rock lobster, tiger flathead and eastern gemfish populations. There are already indications that the peak of lobster puerulus settlement is moving southwards on the east coast of Tasmania and this is expected to continue, to the detriment of settlement to the north. Future conditions are expected to lead to an overall decrease in the abundance and biomass of these three case study species. An additional risk to the lobster grounds in the future as a result of the warmer conditions is the southward spread of the urchin, *Centrostephanus,* which has the potential to form urchin barrens on the grounds. Furthermore, the warm water might also degrade the refuge habitat of species such as giant kelp and associated assemblages, with unknown but likely negative effects on lobster habitat.

Changes to western rock lobster populations are more predictable. The population is likely to respond to warmer temperatures by faster growth of the juveniles and by migrating animals moving to the offshore breeding grounds at a smaller size and maturing at a smaller size. The impact could be expected to be a loss of potential yield unless there is a change in the minimum size in the future, but the smaller size at maturity could be expected to lead to increased numbers of breeding animals being

protected by the legal minimum size. The predicted weakening of the Leeuwin Current will result on average, in reduced settlement in the future.

The effects of fishing on resilience are made up of a collection of potential causes. Recruitment overfishing, where the exploited stock is harvested at a size and intensity that precludes sufficient eggs being produced to replenish the stock is probably the most direct of the effects. Ecosystem overfishing is when the balance of the ecosystem is altered by fishing. Other negative consequences of fishing are where the ecosystem is subsidized through the discarding of bait and by-catch, where the habitat is damaged by fishing gear, where changes in gear technology over time can mask a decline in population numbers and lastly, where fishing can result in genetic change either through selecting for particular life history traits or through a loss of genetic variation in the population.

Improvements in gear technology over time have been an important factor in fishing down the case study populations and have introduced an element of uncertainty into the catch rates that are used to assess the state of the fisheries. In the trawl fisheries, the methods could be expected to have negatively impacted the grounds and influenced the habitat on which demersal species such as tiger flathead are dependent. Spawning biomass levels are particularly low in the eastern gemfish population and in some parts of the southern rock lobster distribution. Although relationships between stock size and recruitment are difficult to pinpoint, the level of spawning biomass in eastern gemfish is considered to be a factor limiting its recovery and contributing to its lack of resilience. Recent published work suggests that subsidies to the ecosystem through discarded bait probably make a significant contribution to the productivity of the western and southern rock lobster populations.

Genetic effects of fishing is a relatively new area of research and very little is known about what effect fishing may have had on the case study species in this report. Potentially there may have been loss of genetic diversity through fishing in the four case study species, but there is no evidence to either support or discount this possibility. Targeting of older size classes above the legal minimum size, particularly in the lobster fisheries where size selection is knife edge, has the potential to select for particular traits which may not enhance the resilience of the populations, but this has not been examined. Research has shown that small southern (but not western) rock lobster females produce eggs and larvae of lesser quality than those produced by larger (older) individuals and this could impact resilience in this and other fished populations.

Coastal zone development resulting from population growth along the seaboard of the country has the potential to affect resilience through the establishment of structures in the sea and through noise, lighting and chemical runoff.

The introduction of exotic marine species has the potential for significantly impacting native species should they become established. There are numerous examples of marine pests having been unintentionally introduced into areas through for example ballast water and then subsequently impacting the environment. In terms of the case study species in this report, there is an indication that screw shells (*Maoricolpus roseus*) could be impacting the ecosystem in 25-80 m depths along the east coast of Tasmania and northwards through the Bass Strait.

Pathogens have been responsible for major mortalities of species in Australia and elsewhere in the world. Outbreaks of infectious diseases are often associated with anthropogenic disturbance such as eutrophication, overfishing and aquaculture. The warming effects of climate change are considered to have a strong likelihood of increasing pathogen development and survival rates, disease transmission and host susceptibility in the future. There have been no records of virulent diseases in the four case study species covered by this report.

Governance and socio-economic considerations are wide ranging in that they cover many aspects that can impact the resilience of the populations and ecosystems with which they are associated. The report has briefly reviewed for the four case study species, the forms of management control, management plans and reference points, effectiveness of compliance, consultation with stakeholders and more.

It is quite clear that many of the factors that can affect population resilience have the potential to interact with each other and some examples of how this can occur have been presented.

The conclusion is that marine animals are innately resilient, but that there are some general characteristics associated with resilience that can be made from the four populations that have been considered in this report and examples from elsewhere in the world.

Environmental effects are clearly critical to the way that the four case study populations have responded to date and in the way that they are anticipated to respond in terms of resilience to the effects of climate change in the future. Western rock lobster in particular, have been shown to respond in very predictable way to the environmental signals on the Western Australian coast.

The effects of fishing mortality on resilience are important, but not overriding. It is clear that there are other forces acting on a population and the ecosystem of which it is a part, that determine a populations ability to recover. Evidence from other studies suggests that the speed that fishing pressure is reduced on stressed populations is a critical element to their chance of recovery. This suggests that likelihood of recovery is dependent on the population or system not being moved to the point where it is replaced by an identity with different functions, structures and feedbacks.

The life history characteristics of the population are also important. The species considered in this report are all fairly similar in terms of age to maturity, but is clear that animals that grow fast and have short generational turnover times would be likely to be more resilient than those that have long lifespans and that are slow to mature.

Minimizing potential stressors is key to maximizing resilience. This is a particularly important consideration in current times of changing climate, expanding coastal populations and demanding socio-economic expectations.

1. Introduction

The term resilience and the definition and concepts behind it, have been discussed at length by Steven Cork (2010) and will not be repeated here. The object of this contribution to the SoE2011 report is to outline a variety of social and ecological factors that can potentially affect the resilience of temperate marine populations, by using four different Australian fish stocks to illustrate practical concepts and issues behind resilience.

In marine ecosystems, resilience operates at several levels but is most recognizable at the species level, and includes natural population fluctuations. In the big picture, the local loss of a population at a location can simply lead to its replacement by a different species that may perform a similar ecological function. Far more catastrophic, is the potential for change at the ecosystem level, where significant change is likely to be broader in scale and more permanent, possibly resulting in changes to the structure and function of the whole ecosystem. Fortunately however, large-scale changes to ecosystems termed regime shifts, have rarely been reported in Australia.

Ideally, resilience should be considered at the ecosystem rather than the population level, because populations are just one ‘cog’ in an ecosystem engine. However, the problem with focusing a discussion article on factors affecting resilience at the ecosystem level, is that that the multitude of interactions at this level would make such an undertaking complex and beyond the scope of a brief discussion document. By comparison, the amount of information available at the population level, particularly for exploited populations such as have been used in this report, is plentiful and therefore useful in providing illustrative material for the concepts that are outlined.

Deciding on case study examples for this report has been difficult, simply because there are so many populations that could be considered. The four populations that have been chosen are all commercially fished species, chosen because they are common in terms of numbers and biomass and therefore are important within their respective ecosystems and because they are well studied and therefore have a wealth of long-term data which can be drawn on from a human impact point of view. A further motivation for using commercially exploited species rather than less studied or by-catch species, is that recent analyses (e.g. Phillips *et al.* 2010) have broadly concluded that there are insufficient data to inform a full analysis of the cumulative impacts of fishing on non-target species in Australia’s marine ecosystems, and that there is little available evidence to assess the possibility that multiple sources of fishing-based effects has a substantive impact on the resilience of by-catch populations or their related ecosystems.

This report is written in two parts. A short section is dedicated to briefly describing the distribution and life history of, and the fishery for, the four commercial fish stocks that are used as case studies in this report. This is not intended to be comprehensive and should be seen as an introductory in its context. Where necessary, some of this information will be expanded later into the document when more detail is provided about factors affecting resilience, in particular when the impacts of fishing are discussed.

The main part of the document deals with each factor separately and describes how a particular factor can affect resilience and where there is evidence in the case study populations, the way that they have been affected and their responses. As will become apparent, there are a multitude of potential factors affecting resilience in fish populations. In many cases these factors interact with each other and resilience, or lack of resilience, cannot be ascribed to a single factor. The possibility that the resilience of a population is responding to a combination of factors needs to be borne in mind by the reader as it is both difficult to overemphasize this reality and to do sufficient justice in pointing out how the different factors may be linked.

2. Background to the case study populations

2.1. Western rock lobsters (Panulirus cygnus)

2.1.1. Distribution

Western rock lobsters are endemic to Western Australia, occurring from Exmouth in the north to Albany in the south. The commercially fished part of the population covers a lesser area stretching from Shark Bay south to Augusta.

2.1.2. Life history

The species is long-lived, with a potential lifespan of 20 to 30 years and a maximum weight in excess of 5 kg. The mean and maximum sizes of lobsters in the population are smaller in the north than the south of the distribution range.

Mature females produce eggs externally on the tail in spring and early summer (Fig. 1). Young females produce only a single batch of eggs, but older (larger) females can produce more eggs and often produce a second batch of eggs within a season. Hatching of the eggs takes place after one to two months and the larvae are swept offshore. The species has a long larval life of around nine months and in this time the larvae move many hundreds of kilometres offshore and undergo several instars as they develop and increase in size. In late winter and early spring the larvae find their way back to the coast, where at the shelf break they metamorphose into what is known as a puerulus, a small colourless lobster-like creature, that swims tens of kilometres inshore and settles amongst rocky reefs and sea grass, mostly in depths of less than 20 m. The puerulus moults to a juvenile shortly after settlement. Three to four years after settlement, many of the juveniles leave the coastal reefs and undertake an offshore migration that leads them to settle in deeper water, up to 100 m. Maturity is considered to occur five years after settlement (Fig. 1).

Juvenile and adult lobsters are opportunistic omnivores, scavenging a wide range of food items. Lobsters themselves are predated on by several different fish and shark species, as well as octopus and even their own species under certain circumstances.

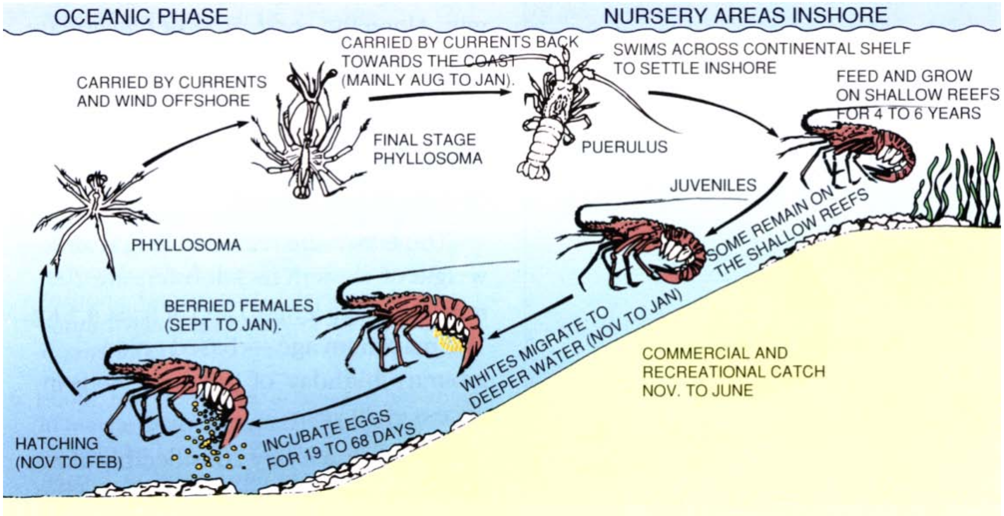

Fig. 1 Diagrammatic life cycle of the western rock lobster

2.1.3 Description of the fishery

Commercial fishing for western rock lobsters began in the 1940s and by the 1950s had expanded to catches in excess of 8,000 t (Fig. 2). Catches are highly variable from year-to-year, given that the fishery is very dependent on just a couple of year-classes and that recruitment is erratic. The fishery has until recently been managed by effort controls, but in the 2010/11 season has moved to individual transferable quotas (ITQs). The fishery is highly regulated and management measures which operate across different management zones in the fishery, include closed seasons, a legal minimum size, legal maximum size for females only, protection of breeding females and effort restrictions limiting the size and shape of pots, the number that can be used and within the season, days that they can be set. Enforcement is rigorous and by and large, is rigidly adhered to by the fishing community.

The western rock lobster fishery is unusual in that research has the capability of being able to predict future safe catch levels based on the monitored level of puerulus settlement in any one year. Monthly settlement monitoring provides an estimate of the number of animals that can be safely caught three to four years after the pueruli have settled.

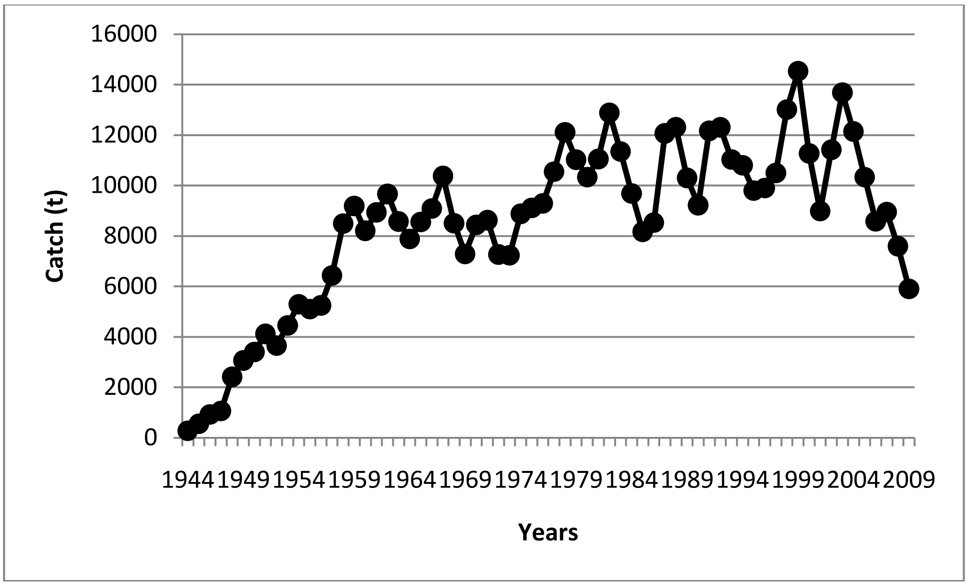

Fig. 2. Catch and fishing effort for the western rock lobster fishery from 1944 onwards (data from de Lestang, Department of Fisheries Western Australia, pers. comm.).

2.2. Southern rock lobsters (Jasus edwardsii)

2.2.1 Distribution

Southern rock lobsters, in commercial quantities, extend along the south coast of Australia from around Augusta in Western Australia to the southern coast of New South Wales and south to Tasmania. The same species is also found in New Zealand waters where it also supports a large commercial fishery.

2.2.2. Life History

As with western rock lobsters, southern rock lobsters are long-lived and have a similar life history to that shown for western rock lobsters in Fig. 1. The eggs hatch after being carried by the female for three to five months over winter and spring (Frusher *et al*. 2000). Only one batch of eggs is produced within a season, but older (larger) individuals produce many more eggs per batch than young (small) lobsters. The larval phase lasts 16 to 23 months during which time they may move many hundreds of kilometres from the coast and moult through 11 morphologically distinct stages (Frusher *et al*. 2000). The final stage phyllosoma metamorphoses to a puerulus adjacent to the continental shelf and swims inshore to settle on the reefs (Frusher *et al*. 2000). Unlike western rock lobsters, southern rock lobsters do not undertake a migratory phase in the life history. Juveniles mature at around five years after settlement (Frusher *et al*. 2000).

2.2.3 Description of the fishery

Southern rock lobsters were fished for domestic consumption in the late 1800s, but commercial exploitation really only began in earnest in the late 1940s with freezer shipments to the USA (Phillips *et al*. 2000). Each of the four southern states with commercially important southern rock lobster fisheries manages their lobster resource slight differently to the others (e.g. there are some differences in the legal minimum size across the fishery). At one stage all of the southern rock lobster fisheries operated under input controls, but over time, all but the Western Australian fishery have moved to output controls. The first one to go to a quota system was the southern zone in South Australia in the 1993/94 season.

Southern rock lobster catches in the different states and management zones within those states have fluctuated over time (Fig. 2), in part due to the application of different management measures coming into force in different areas at different times. However, in the last few years there has been a sharp and as yet not fully explained decrease in landings across the fishery (Linanne *et al*. 2010).

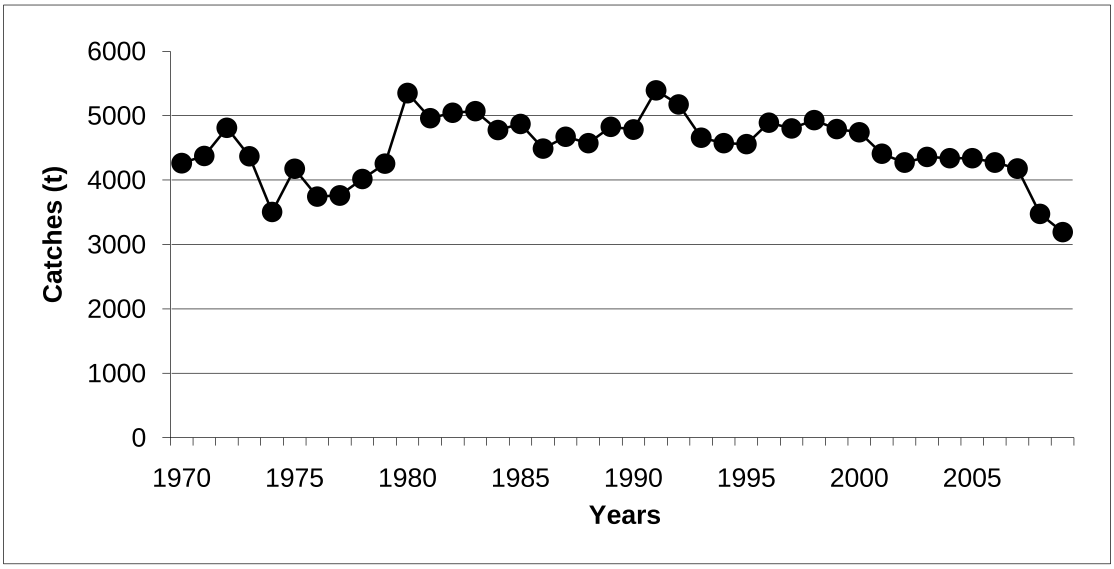

Fig. 3. Catch and fishing effort for the southern rock lobster fishery in Tasmania, Victoria, South Australia and Western Australia from 1970 onwards. Landings for years prior to 1970 for the full fishery (i.e. all states) are unreliable and incomplete. (Data for South Australia, Tasmania, Victoria and South Australia from Linanne, SARDI Aquatic Sciences, pers. comm., data for Western Australia from How, Department of Fisheries Western Australia, pers. comm.).

2.3. Tiger flathead (Neoplatycephalus richardsoni)

2.3.1. Distribution

Tiger flathead are endemic to Australian waters and are distributed from Coffs Harbour, New South Wales, to Portland, Victoria, including Bass Strait and Tasmania (Kailola 1993).

The bulk of the tiger flathead catch is taken offshore by the SESSF, a Commonwealth managed fishery. The balance of the catch which is taken inshore, is managed by the NSW, Victoria and Tasmanian state fisheries departments.

2.3.2. Life History

The species has an extended spawning period from spring through to summer in New South Wales and the Bass Strait (Kailola 1993). The fish tend to move to the shallower inshore waters to spawn and as well they tend to aggregate into shoaling groups at this time, which leads to higher catch rates in the spawning season (Tilzey1994). Egg production is up to 2.5 million per individual and the larvae are thought to be pelagic. Females reach maturity at 4 to 5 years and live to around 12 years and a maximum length of 64 mm; males by comparison live 8 to 10 years and because they are slower growing than the females, they seldom reach over 50 cm (Kailola 1993). Tiger flathead leave the seabed at night to feed on animals in the water column; juveniles feed mainly on crustaceans, while adults feed mostly on small fish (Kailola 1993). There is little evidence for migration by this species (Tilzey 1994).

2.3.3. Description of the fishery

According to Klaer (2010), tiger flathead have been commercially fished since the development of the steam trawl fishery in 1915. Steam trawlers were used up until about 1960. Danish seine gear, a fishing method which is still being used today, developed in the 1930's. Diesel trawlers began landing tiger flathead in the 1970's and currently this and Danish seine methods take the total catch (Klaer 2010). A TAC was introduced in the South East Fishery for this species in 1992

Landings first peaked in the late 1920s at over 3,700 t, before dropping back in the 1930s and 1940s to very low levels of less than 500 t (Fig. 4). Catches rose again in the 1950s, peaking in the mid-1960s at over 3,700 t before declining to around 1,500 t for several years in the mid to late 1970s (Fig. 4). The most recent peak catches have been 1999 through 2003, at over 3,600 t. Landings in the last few years have been trending downwards (Fig. 4).

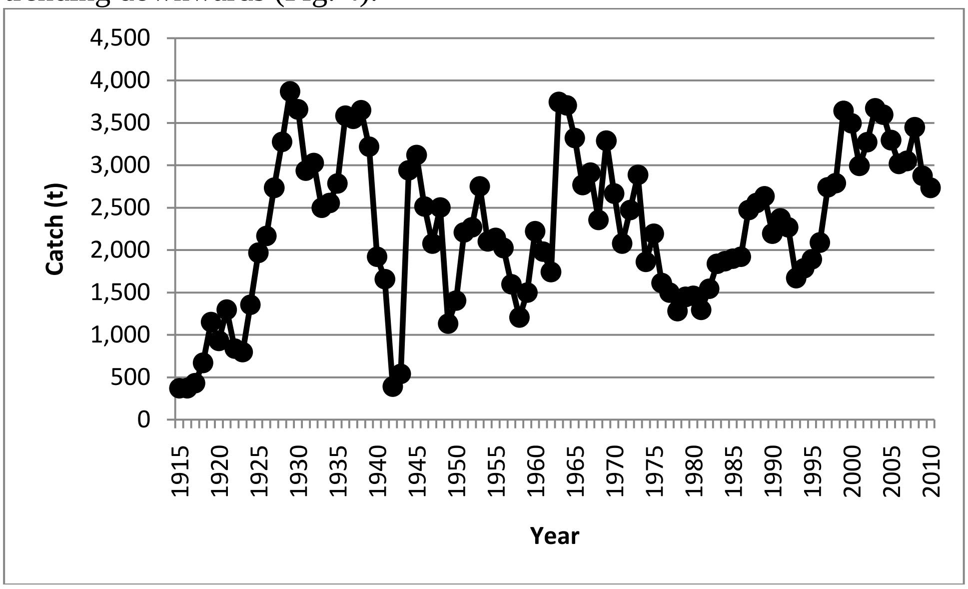

Fig. 4 Total retained catches of tiger flathead 1915-2010 calendar years, with estimated 2010 catch (data from Klaer, CSIRO, pers. comm.).

2.4 Eastern gemfish (Rexea solandri)

2.4.1 Distribution

Gemfish have a much wider distribution than tiger flathead. They occur from Cape Morton in southern Queensland, to the central Western Australian coast (Kailola 1993).

Two genetically distinct stocks are recognised across southern Australia – an eastern Australian stock which is the one discussed here and a southern western stock (Paxton and Colgen 1993, quoted by Wilson *et al*. 2009). Some overlapping of the two stocks occurs off western Tasmania.

2.4.2. Life History

Gemfish occur in depths of 100-700 m in the deeper continental shelf and upper slope waters (Kailola 1993). Although they are a mid-water species, they are vulnerable to demersal trawling, particularly when the mature adults aggregate and migrate along the shelf break off southern New South Wales (Prince and Griffin 2001). Spawning grounds are off the central to northern NSW coast and the annual spawning occurs in August (Kailola 1993). Females produce 1-2 million eggs (Kailola 1993). Growth is relatively fast, with males maturing at 3-5 years and females at 4-6 years. Maximum age is considered to be ~11 years for males and 16 years for females (Kailola 1993). The species is carnivorous, feeding mainly on deepwater fish, but also on squid and crustaceans (Kailola 1993).

2.4.3 Description of the fishery

The demersal trawl fishery for eastern gemfish began in the late 1960s along the Sydney to Newcastle stretch of coastline in the winter spawning aggregations (Prince and Griffin (2001). The fishery expanded rapidly and by the late 1970s was landing ~5000 t per annum (Fig. 3).

Rowling (2001) states that prior to 1988 fishing was virtually unregulated and annual catches were exceeding 3000 t per annum (Fig. 5). However, declining catch rates and sizes of fish in the catch led to the introduction of a TAC for gemfish in that year and this was followed in 1989 by the introduction of ITQs for the species. Rowling (2001) identifies a run of poor cohorts recruiting to the brood stock between 1989 and 1992 as the precursor to a rapid decline in landings to <1000 t (Fig. 5).

In response to concerns about the state of the stock, a zero TAC was introduced between 1993 and 2000, except for 1997 when a 1000 t TAC was set only for the one year (Rowling 2001) Since 2002, the fishery has had a zero recommended biological catch (RBC) and an annual by-catch of 100 t (Wilson *et al*. 2009).

Model estimates suggest that recruitment was variable but high up until the mid-1980s, but apart from a strong cohort spawned in 2002 (although weak by comparison with those prior to the mid-1980s), recruiting year classes have remained weak (Anon. 2010). The fishery remains severely depleted. It currently operates under a by-catch TAC of 100 t and has been listed under the EPBC Act as “Conservation Dependent” (Anon. 2010).

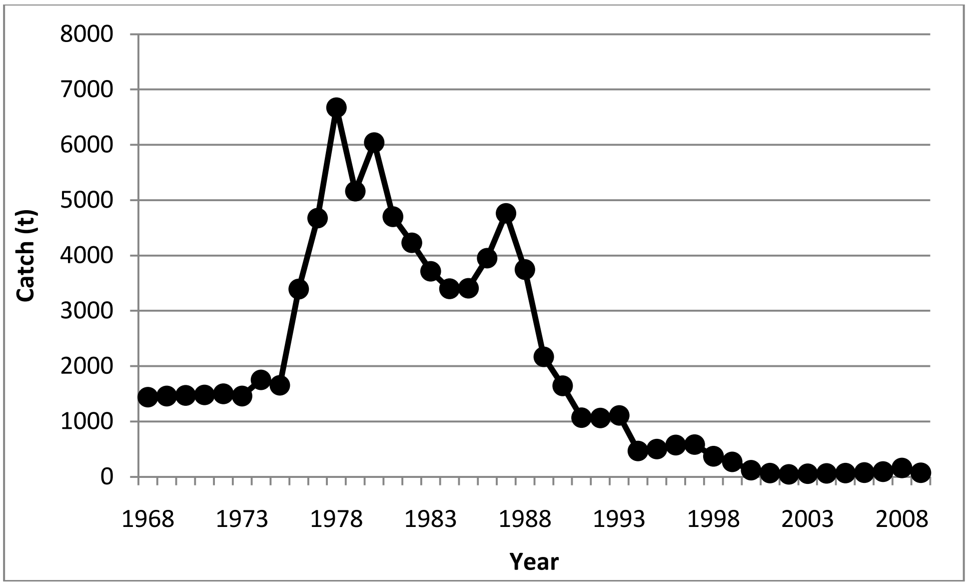

Fig. 5 Total reported retained and by-catch landings of eastern gemfish 1969-2009 calendar years (Little and Rowling 2010).

3. Factors potentially affecting population resilience

Because this report has used exploited species to illustrate how different factors may impact resilience, it is useful to consider at this point how appropriate the relationship is between the data available for commercial landings (which is an estimate of the total catch) and the ecological condition of a population.

The harvested part of a population is generally made up of the larger individuals. The non-targeted ‘under size’ component of the population which is either unaffected or minimally directly affected by fishing, is often a very substantial proportion of the total population biomass. These smaller individuals are frequently immature, sometimes spatially separated from the harvested population and because of their smaller size they would be unlikely to be occupying the same trophic feeding niches as the larger individuals.

Once individuals in a population reach a harvestable size, if it is a well-managed fishery, only a proportion of the harvestable-size animals are removed by fishing each season. While there will be year-to-year variations in catch caused by environmental effects on aspects such as breeding and settlement success, the long term mean catch should theoretically be unchanged relative to the success of breeding, unless it is acted on by a disruptive influence. After the initial fish-down phase of the early years and a fishery has become mature and sustainable, long term trends in the annual landings are therefore, in at least some circumstances, a reasonable, cost-effective, surrogate for estimation of the population size of the unfished stock and its responses to ecological, socio-economic and management forces.

There are a multitude of such factors that have the potential to affect population fluctuations and ultimately resilience of those populations. These factors can act either alone, or in concert with each other. Most of these factors are either directly or indirectly attributable to anthropogenic influences, though there are some cases where human influence is less clear, for example where a pathogen might decimate a population. But even then, anthropogenic stressors may play a role in decreasing the resilience of the population in question.

There are six main categories of factors that have the potential to influence population resilience of fished species; these are climate change, fishing, coastal development, introduced marine pests, infectious diseases and the socio-economic influences associated with fishery management.

3.1. Climate change

There is a wealth of information on the predicted effects of climate change on the Australian marine environment (see for example Hobday *et al.* 2006; 2008, Poloczanska *et al.* 2007). It is anticipated that changes to environmental variables such as ocean temperature, wind, currents, water chemistry rainfall and extreme weather patterns will have direct and indirect influences on marine ecosystems and in some cases changes to these stressors have already been recorded.

Climate change has been considered in the broad sense by Walther *et al.* (2002), as influencing biological systems by modifying (i) their phenology and physiology; (ii) their range and distribution; (iii) the composition and interaction within communities and (iv) the structure and dynamics of communities. The focus in this report is how these modifications brought about by the singular and combined effects of different stressors can affect population resilience.

It needs to be noted that climates are inherently variable. The differences between climate variability and climate change is that variability refers to changes on annual or decadal scales, whereas climate change is used to describe long-term change.

3.1.1. Temperature

According to Poloczanska *et al*. (2007), predictions from CSIRO’s climate model, are that waters around Australia will warm by 1-2^{o}C by the 2030s and 2-3^{o}C by 2070. Furthermore, this pattern of warming is not predicted to be uniform; greatest levels of warming of the marine environment are predicted to occur in the southeastern region of the country. According to Cai *et al*. (2005), the intensification of the East Australian Current will generate a warming rate in the Tasman sea that is predicted to be the largest in the southern hemisphere.

The predicted water temperatures for 2030 and 2070 (see Poloczanska 2007) will result in temperatures in the Northern Tasmanian rock lobster grounds being equivalent to those currently occurring in the waters around the Victoria-New South Wales border and by 2070, the differences will equate to areas even further north than that (Pecl *et al*. 2009). Whilst there are very likely reasons other than temperature that limit population abundances of southern rock lobsters north of the Victorian border, Pecl *et al*. (2009) consider that warmer water may result in lower abundances of lobsters on the northern Tasmanian grounds into the future.

One possible reason put forward by Pecl *et al*. (2009) for lower abundance of southern rock lobsters in the warmer areas of the distribution range, is that water above 21.5^{o}C has been shown under laboratory conditions to result in total mortality of larvae (Bermudes and Ritar (2008). Average temperatures in Tasmanian waters in 2030 are predicted to be above 20^{o}C during part of the year in some northern areas of the fishery by 2030 and across even more months by 2070 (Pecl *et al*. 2009).

Growth increments and size at maturity are considerably larger in the north of Tasmania than the south and temperature has been strongly implicated in these observed differences (Gardner *et al*. 2006) [note southern rock lobster grow larger in warmer waters, whereas western rock lobsters have a smaller maximum size in warmer water]. In modelling work predicting trajectories of southern rock lobster exploitable biomass off the Tasmanian coast under climate change scenarios (Pecl *et al*. 2009), the increased growth as a result of warmer water drives future increases in biomass. The increase in biomass is not sustained due to predicted changes in puerulus recruitment on that coast (but this is dealt with later – see 3.1.2.).

Naturally there are many issues that can not be taken into account by the modelling work. It is likely for example that warmer water will lead to increased predation, and therefore higher mortality rates than is presently the case. Also the urchin, *Centrostephanus,* would be likely to establish itself more widely in warmer water. Without intervention, this could lead the formation of urchin barrens (Ling 2008), which are areas where macro-algae can be decimated by excessive urchin grazing leading to lower productivity and fewer lobsters.

While the Tasmanian part of the southern rock lobster fishery may be resilient to the substantial increases in temperature predicted across most of their fishery, the same cannot necessarily be said for the grounds to the north stretching from Western Australia through South Australia and Victoria. There is predicted (Lough 2009) to be slightly less warming of the sea surface temperature in the southern part of the continent than the 1^{o}C rise predicted by 2030 in most other regions. A lesser rise compared to the rest of the continent is also predicted for the southern region in 2070 (1.5-2^{o}C) in 2100 (Lough 2009).

The same reason that Pecl *et al*. (2009) considered that temperature may affect southern rock lobster abundances on the northern Tasmanian grounds in the future, may apply to populations along the southern coast of the mainland. Based on commercial landings, the southern zone of South Australia would appear to be the most productive region for southern rock lobster (Phillips *et al*. 2000). If the apparently lower densities to the west and east of this region are due to the warmer waters in both directions, then it would be logical to assume that the population distribution will contract from Western Australia and Victoria towards South Australia in the future. As in the Tasmanian situation, in those areas where southern rock lobster continue to flourish in the future, growth rates could be expected to respond positively as temperature increases.

Temperature changes in these southern areas might also degrade the remaining refuge habitat of species such as giant kelp and associated assemblages (Okey *et al*. 2006), with unknown but likely negative effects to lobster habitat.

Of the two fish populations being considered here, temperature might be expected to have more influence on tiger flathead which are demersal and found shallower than eastern gemfish, a mid-water species with a much deeper water distribution range. As noted earlier tiger flathead move inshore into shallow waters to spawn and since under climate change scenarios shallow waters would probably experience the largest temperature change, it may be this stage in the life cycle that leads to the core of the population moving southwards. Poleward distribution shifts ascribed to raised water temperature have been recorded in some species (e.g. Perry *et al*. 2005) and many more are expected to respond in the same way as temperatures exceed their physiological thresholds (Rosenweig *et al*. 2008). It would seem unlikely that temperature would have had impact on resilience of the two species to date, but in the case of tiger flathead this may be a possibility in the future if the population is driven southwards and coastal habitat is less extensive than it is in the present distribution range.

The south western marine regions are also expected to experience high levels of warming relative to other parts of Australia and indeed this region has already recorded an increase of 1^{o}C since the 1950s (Pearce and Feng 2007), with most of the increase having occurred in the cool autumn-winter months of April through September (Caputi *et al*. 2009).

In the case of western rock lobsters, changes in ocean temperature that have been experienced are considered to be a factor in the decrease at the size at which the lobsters are maturing now compared to the mid-1970s (Melville-Smith and de Lestang 2006). Furthermore, it is known that juvenile western rock lobsters grow faster in warm water (Johnston *et al*. 2008) and that they migrate to the offshore breeding grounds at a smaller size if the temperature was warm at the time of their settlement (Caputi *et al*. 2010a). These relationships between temperature and the biology of the animal shows the importance of water temperature in driving changing patterns of exploitation over the life cycle of the animal. Predicted temperature increases would be expected to exaggerate those changes further in the future.

The legal minimum size for western rock lobsters is below the size at first maturity in the southern part of its distribution and above size at maturity to the north. One outcome from the decreasing size at maturity that has been recorded is that lobsters will increasingly be maturing below the legal minimum size over more of their distributional range each year. Since mature animals grow slower they will therefore not be available to the fishery as quickly as is currently the case. Slower entry to the fishery means that there will be greater opportunity for potential yield to the fishery to be lost to natural mortality and without any change to the legal minimum size, it is likely that this will lead to a decrease in productivity in the fishery in the future. Nonetheless, this will increase the availability of the smaller rock lobsters to natural predators, and increase their value in ecological function terms, and possibly provide for enhanced resilience of species that are dependent on the rock lobster as prey.

Fortunately, western rock lobsters have a wide temperature tolerance. Chittleborough (1975) for example, ran long-term experiments with animals held up to 29^{o}C and for short periods up to 34^{o}C, temperatures that were well above the temperature range (15.8-27.6^{o}C) he recorded in the central part of the fishery at Seven Mile Beach.

From that work he determined the optimal temperature for western rock lobsters in terms of growth and survival to be 25-26^{o}C. Responses by western rock lobsters to the temperature increases that have been observed to date would point to the population being relatively resilient to this particular stressor.

According to Feng *et al*. (2009), the warming trend that has been recorded off the west coast over the past five decades will continue, with sea surface temperatures projected to be 0.7 – 1.1^{o}C warmer by 2030 and 2-2.6^{o}C by 2100. On the positive side, the smaller size at maturity that will likely result from this increase in temperature could be expected to increase egg production due to more mature females being protected by the legal minimum size which may add to the enhancement of the resilience of the population and dependent predators on the various life stages.

3.1.2. Currents

Future predictions are that the East Australian Current (EAC) will strengthen and extend further south than is presently the case. On the West coast, the Leeuwin Current is not expected to strengthen, but to the south, there is expected to be greater westward transport of water mass in the Great Australian Bight region.

Changes to the EAC are what will be responsible for driving the exceptionally large increase in water temperature that is predicted will occur in the south eastern region of the country in future years (Cai *et al*. 2005). The impact that this increase in temperature is likely to have on the case study fisheries has already been discussed. Outside of influencing water temperature, the other area that ocean currents could be expected to impact in terms of the case study species, would be in the transport of fish eggs and fish and lobster larvae as well as interactions with oceanic and topographical features that leads to regional differences and seasonal changes in productivity driving the ecosystem.

The major sources of nutrients to the South East trawl fishery comes from the mixing of the Sub-Antarctic Waters with the EAC to the east of Tasmania and the Leeuwin Current to the West (Prince 2001). The nutrient rich waters are transported onto the shelf in a variety of ways, in some cases deepwater currents playing a part, in other cases wind driven water, topographical features and complex overturning of surface water bringing the deeper nutrient rich water to the surface and onto the shelf (Prince 2001). It is believed (Prince 2001), that these productivity events are what drive feeding and breeding aggregation events, and that these in turn are responsible for the seasonal differences in catchability and availability of the shelf-break species in the south east of the country. Eastern gemfish aggregations are believed for example, to be in response to these type of events (Prince and Griffin 2001).

How will these EAC changes affect eastern gemfish and tiger flathead? Okey and Hobday (2006) consider that the projected climate change effects in the south east, of general warming, combined with a stronger flowing EAC, will lead to an overall decrease in the abundance, biomass, productivity and diversity of benthic and demersal fish in the region.

There has been a multi-decadal weakening of Leeuwin Current strength since the mid-1970s and according to model predictions (Feng *et al*. 2009) that trend will continue into the future.

The size of the Western rock lobster settlement is well known to be highly correlated with the strength of the Leeuwin Current (Caputi *et al*. 2001), with low puerulus settlement being a feature of years when the Leeuwin Current is flowing slower and this in turn leading smaller catches three to four years later when that age-class becomes available to the fishery.

The spatial distribution of settlement is also highly correlated with the strength of the Leeuwin Current, with the peak of settlement occurring up to 2^{o} of latitude further south in years when the current is strong compared to those when it is weak (Caputi 2008). The implication of this in terms of resilience of the population, is that the numbers of animals and the extent of their distribution are likely to be reduced under the predicted continued weakening of the Current strength.

To the south of Australia, transport of southern rock lobster larvae over their lengthy larval life is influenced by the Leeuwin Current in the west and the EAC in the east and within those bounds, the complex zonal flows off southern Australia and the low energy eddy field south of the Great Australian Bight (Bruce *et al*. 2007). Relationships between these water movements and the strength of puerulus settlements are not as clear as Leeuwin Current strength is for western rock lobster settlement. It is therefore not clear how southern rock lobster settlement on the mainland will be affected by future predictions of changes in Leeuwin Current strength.

On the east coast of Tasmania, puerulus settlement appears to be influenced by the position of the sub-tropical convergence, where the EAC meets the cooler Southern Ocean water. Pecl *et al*. (2009) has shown a trend in puerulus settlement data collected since the early 1990s, which suggests that settlement levels at Bicheno in the north eastern part of the state have been declining since the mid-1990s, those at Iron Pot further south have been declining since the mid to late 1990s, while those in the south of the state at Recherche Bay were low until the mid-1990s and have trended upwards since then. Over the same period, the EAC has been penetrating further south.

The implications of this are that with the EAC flowing stronger and extending further south in the future, that peak settlement will continue to track southwards. Pecl *et al*. (2009) have used different puerulus settlement scenarios, and future TACs compared to current TAC settings, to assess how the abundance and distribution of southern rock lobsters may alter in the Tasmanian fishery as a result of climate change. As expected, given the different assumptions and the sensitivity test applied to the model projections, results were variable. However, overall they predicted initial gains due to improved growth rates associated with warmer water (discussed in 3.2.1), followed by a reduction in biomass as the expected decline in recruitment due to the declining settlement in the north affects population numbers.

3.1.3. Wind

Some of the interactions between wind and ocean currents have already been noted in the previous paragraphs; since wind is a driver of currents, the two are difficult to view in isolation.

It has been predicted under climate models that there will be a southward shift in the zonal winds that normally cross the southern part of Australia and that this will lead to an overall weakening of the winds over the southern portion of the country (Gillet and Thompson quoted by Okey and Hobday 2006, Cai *et al*. 2005).

It is known that strong westerly winds drive cold, nutrient rich sub-Antarctic waters up the east coast of Tasmania in spring and summer and in years with frequent windy periods, the high productivity experienced in the spring blooms can continue through summer, positively influencing production in the system (Harris *et al*. 1988). The predicted weakening of the winds could be expected to result in reduced productivity in the southern region of the country, which will in turn impact on the survival and growth of fish stocks on this part of the coast.

In terms of the fish populations, Thresher (1994) has pointed to a correlation between annual wind strength and gemfish year-class strength. That relationship indicated that years that record unusually large numbers of days of strong westerly winds, corresponded with years of good gemfish recruitment.

Similarly, zonal westerly wind has been shown in Victoria to be positively correlated with larval abundance of King George whiting in the season of spawning and with catch rates three to five years later when the species recruits into the commercial fishery (Jenkins 2005). It would not be unreasonable to consider that as with King George whiting, the productivity of tiger flathead, which is also a demersal inshore species, might be related to zonal westerly wind strength.

Wind strength and direction would be expected to be important in creating conditions that could assist southern rock lobster pueruli in recruiting to the coast. Although seasonal wind directions that would promote downwelling and therefore onshore transport of water correspond with the peak winter settlement off the South Australian coast, Bruce *et al*. (2007) were unable to show any correlation in their oceanographic-lobster larval modelling work, between wind strength and the corresponding strength of the puerulus settlement. The same authors noted that the fact that Eastern Tasmania does not have a strong annual cycle in downwelling-favourable windstress may explain the lower seasonal signal in puerulus settlement on that coast compared to the South Australian coast. Overall therefore, Bruce *et al*. (2007) have concluded that there is no simple relationship between wind strength and direction and that therefore wind is unlikely to be the primary cause of observed seasonal and interannual differences in settlement variability.

In terms of western rock lobster, there is a clearer relationship between wind and puerulus settlement strength. Caputi *et al*. (2001) used rainfall in winter and spring July-November) as a proxy for storm driven westerly onshore winds. The resulting positive correlation suggests that the strong winds which would drive near-surface currents onshore, may assist pueruli in getting from the shelf break where they first become pueruli, to inshore areas where they settle. One of several changes that Caputi *et al*. (2010a) consider may occur under climate change, is the weakening of westerly winds in winter; by implication this could be expected to negatively affect settlement of western rock lobsters in the future.

3.1.4 Ocean acidification

The rise in CO_2 levels as a result of fossil fuel emissions is leading to an increase in acidity and a decrease in the calcium carbonate saturation state of the world's oceans as they absorb CO_2 from the atmosphere (Matear, 2006). The concern relating to these changes is that many marine animals from corals through to molluscs and crustaceans are dependent on external calcium carbonate-based skeletal structures and changes in the availability of calcium carbonate may affect their shell structures and ultimately their populations and productivity.

Orr *et al*. (2005) have modelled the ocean carbon cycle to assess calcium carbonate saturation in the future. Their results, based on a 'business as usual' scenario for future CO_2 emissions, suggest that within 50 to 100 years southern ocean surface waters will reach saturation levels that are sufficiently low in aragonite (a form of calcium carbonate) to affect growth of the protective shells of pteropods (a planktonic organism). The implication is that these changes in sea water chemistry could have dire consequences for many other calcifying organisms over the next century.

As crustaceans with a calcified exoskeleton, western and southern rock lobsters would be amongst the organisms that could potentially be affected by future aragonite saturation levels. In terms of fish, these future predicted changes could impact their food because many species of plankton, the basis of the foodchain, have calcareous structures. There is no suggestion that current levels of acidification are having effects, but clearly these changes would have the potential to disrupt the resilience of ecosystems and the communities within them.

3.1.5. El Niño Southern Oscillation (ENSO) events

ENSO has a major influence on Australia's interannual climatic variations and as such, it affects some parameters that have already been discussed (currents, wind strength and direction and temperature and their affects on the case study species) as well as many others that are less relevant to this report (e.g. rainfall and cyclones).

In the marine environment of the West coast, El Niño years are characterised by a slower flowing Leeuwin Current. This results in more intense drought periods which would mean fewer winter storm fronts on the west coast, with less onshore wind associated with those fronts and consequently reduced settlement of western rock lobster pueruli.

The influence of ENSO on the East Australian Current is weaker than it is on the Leeuwin Current and observational evidence on how the EAC varies with ENSO is limited (Holbrook *et al*. 2009). The projected increases in temperature on the South East coast caused in a large part through the intensification and southward extension of the EAC, could be expected to be enhanced by warm air temperatures associated with El Niño events.

According to Holbrook *et al.* (2009), model simulations are suggesting that climate conditions will be moving towards an 'El Niño like' state in the future, but there is no consensus about the frequency or amplitude of ENSO events. Effectively this means that there is a high degree of uncertainty surrounding the effects of changes in ENSO variability on Australia's marine environment related to climate change.

3.2. Fishing

Most scientists would probably accept that an extreme shock to an ecosystem could lead to many years of recovery and would be unlikely to see the system "bounce back" to exactly the same state or with similar types and levels of ecological function that prevailed before the shock. However, in fisheries science high yields and high exploitation rates through mismanagement are accepted as being entirely reversible without adverse effects under a more conservative management regime, provided that the ecosystem itself, including its functional aspects, has not shifted as a result of the exploited stock (or stocks) having been overfished. Despite this optimism, there is a litany of examples in the international fisheries literature showing resources fished beyond their sustainable limits that have not been sufficiently resilient to recover in the short to medium term (e.g. monk seals, pearl oysters and two lobster species (Schultz 2011) and closer to home, eastern gemfish – Little and Rowling 2010). Stocks that have recovered from overfishing (e.g. Fig. 3e, in the meta-analysis by Worm *et al.* 2006), are the exception rather than the rule.

Fishing and fishing activities can affect target stocks in a number of different ways. These include growth overfishing, recruitment overfishing, so-called ecosystem overfishing, and others.

3.2.1. Growth overfishing is when the exploited stock is harvested at a size or weight below which the optimal yield per recruit can be achieved. What this means is that fishing pressure, or harvest rates, are at such a level that the fish are caught while they are still in the fast growth phase of their life cycle – usually this means targeting of younger animals before they have reached maturity, because for most animals the rate of growth slows down after maturity is attained. While growth overfishing is important from an economic consideration, it does not by itself affect sustainability of fishing of a stock and therefore by implication, might be considered to be of no consequence to the fishing resilience of a population when fishing is undertaken within a good management regime.

3.2.2. Recruitment overfishing is when the exploited stock is harvested and reduced to a size and with a fishing intensity that precludes sufficient eggs being produced by spawning animals, to replenish the stock.

In the case of western rock lobsters, the fishery has high exploitation rates and has a legal minimum size that for most parts of the fishery harvests animals that are around the size at which they achieve maturity. This type of situation could potentially lead to recruitment overfishing and the managers have always been well aware of that possibility. To avoid this, care has always been taken to manage egg production in the fishery by attempting to keep egg production above levels measured in 1980 (Phillips *et al.* 2007). That year was chosen as a target because at that point in time there was little concern from scientists about the amount of egg production and there was certainly no indication then that egg production was inadequate for the purposes of maintaining the fishery. Subsequently egg production did decline below 1980 levels and by the early 1990s there was evidence that recruitment in the fishery was being impacted at the offshore Abrolhos Islands region of the fishery (Caputi *et al.* 2005). This led to management measures being introduced in the early 1990s to rebuild the breeding stock (since a reduction in breeding stock was considered the most likely cause of the decline in recruitment) and since the mid-1990s egg production has consistently been measured in the fishery as being over and above the 1980 level. In terms of resilience, there was no further evidence that spawning stock of the lobster was a significant factor affecting the level of settlement, which suggests that either the assumption about cause of recruitment decline was incorrect or that the management measures aimed at improving the brood stock were successful.

In recent times, since the 2006/07 season, there has been a further downturn in recruitment which has failed to be fully explained. A risk assessment workshop held in 2009 to review the cause of the downturn (Brown 2009), considered that it could have been caused by either changes in environmental conditions and productivity in the Eastern Indian Ocean, or a decline in the breeding stock, particularly egg production in the northern part of the fishery which has been thought to be (from oceanographic modelling) particularly important in terms of producing larvae that are successful in returning to the coast (Caputi *et al.* 2010b). How resilient the fishery and the stock will be to this latest recruitment downturn remains to be seen.

The status of egg production in the southern rock lobster population is very different across its distribution. Egg production in Tasmania for example, is considered to be above 80% of the pristine level in the southern areas (Phillips *et al.* 2010) where growth is slow and where much of the breeding stock is protected by the legal minimum size, but around 15% of pristine levels in the northern areas (Phillips *et al.* 2010) where growth is fast and where the legal minimum size offers less protection to breeders. In Victoria, egg production is considered to be ~47% in the Western Zone and ~30% in the Eastern Zone, while in South Australia both management zones had estimates of egg production in 2007/08 of 10-15% of pristine levels (Phillips *et al.* 2010).

It appears that there is no magical figure that defines the critical point below which the level of egg production affects future levels of settlement in a population. As previously mentioned, there was evidence that western rock lobster recruitment was being impacted in part of the fishery in the early 1990s. At that point modelling results were suggesting that egg production levels were 15-20% of pristine (Anon. 1993). Based on this experience, there would have to be concern about current egg production levels in parts of the southern rock lobster fishery. However, there is currently no indication of low recruitment – South Australia and Victoria received some of the highest puerulus settlements on record in 2005, 2006 and 2007 (Linnane *et al.* 2010).

Not only is the question of how many eggs is enough difficult to answer, but it is complicated in the southern rock lobster population by the contribution that is made to settlement in one region by larvae spawned in another. All regions, except Western Australia (which has a very small southern rock lobster population), are considered to receive more recruitment from outside their own boundaries than from self-recruitment (Bruce *et al*. 2007).

In summary, while there is no evidence that the southern rock lobster population is being affected by recruitment levels, low levels of egg production in some areas should be of concern. The above discussion however also infers that knowledge of the processes that link estimates of breeding stock and egg production to subsequent settlement and rock lobster population sizes are highly uncertain, and hence can only be used to weakly infer population resilience.

Spawning biomass levels for eastern gemfish are believed from modelling outputs, to be at ~15% of the unfished situation, which in the case of the model is 1968 (Little and Rowling 2010). The same modelling work suggests that spawning biomass declined rapidly through the 1980s and early 1990s. It stabilised at very low levels during the early to mid-2000s and now shows signs of recovery to levels last seen in the 1990s. It is noted by Little and Rowling (2010) that there is the potential for catch rates in this fishery to be 'hyperstable', meaning that catch per unit effort could be over estimated because of the shoaling behaviour of the fish. If this was this the case, it would have the effect of producing an overly optimistic state of the spawning biomass.

As breeding stocks and egg production decline towards zero (in situations of overfishing) recruitment of juveniles into a fishery will be reduced, perhaps dropping precipitously after a minimum threshold has been breached. As mentioned above, there is no standard 'safe' level of egg production relative to the unfished state, below which recruitment can be confidently expected to be impacted, so representing a generic threshold for fishery managers to avoid. However, meta analyses of stock recruitment data from exploited fish stocks have been used to examine threshold levels of egg production relative to unfished levels beyond which catches are affected (Mace and Sissenwine 1991). Mace and Sissenwine's (1991) analysis showed that threshold levels are strongly influenced by the taxonomic affiliation of the fish species as well as certain life history parameters. That aside though, in 80% of the stocks they considered, maintaining egg production above 30% of the pristine unfished situation was enough to ensure stock replacement for fisheries resilience. So, at least for the purposes of maintaining a level of fisheries production, 30% of pristine egg production might be considered to be a generic low threshold to be well avoided by fishery managers. It remains unclear how this threshold for fisheries production might relate to the ecological resilience of the population (such as might become important in responding to other pressures such as climate change).

The level of spawning biomass in eastern gemfish is far below 30% of its unfished spawning biomass, which would suggest that this may be one of the factors limiting its recovery and contributing to its lack of resilience.

Spawning biomass levels for tiger flathead are currently estimated to be ~44% of the unfished state (Klaer 2010). That modelling work shows that the spawning biomass depleted rapidly from the start of the fishery in 1915 and bottomed out in the 1950s to around 20% of pristine. In the 1970s it recovered and by the 1990s reached levels of around 40% that are still being maintained today.

3.2.3. Ecosystem overfishing is when the balance of an ecosystem is altered by fishing. The term 'fishing down the foodweb' is a catch phrase that is commonly used in the popular literature describing one scenario of ecosystem overfishing in which the larger predatory fish, the apex predators in an ecosystem, are removed by fishing because they are generally the ones most sought after by the markets and the ecosystem then shifts towards one dominated by small lower order consumers – see Ward and Myers's (2005) example of changes in the pelagic fish communities of the tropical pacific. In intensively fished areas of the world, not only have the top order predators been removed, but so too have smaller fish and invertebrate species, leading to shifts in the structure of the ecosystems.

Changes such as those above are extreme outcomes, and there is little published evidence of such serious detrimental impacts to the ecosystems of which the four species used in this case study belong. However, this does not necessarily imply that these changes have not occurred, but probably more that historical information is inadequate to enable an effective assessment to be conducted. One study (Graham *et al*. 2001) showed that the biomass of sharks and rays on the upper continental slope of the trawling grounds in the SET fishery had declined drastically over a 20-year period between 1976 and 1996 and had reached very low levels by 2006.

While longitudinal studies (studies through long periods of time) of changes in trophic levels Australian ecosystems are limited, there are studies reporting on fishing induced changes to the ecosystems of which the species being considered by this case study are a part, through productivity enhancement by bait and also to habitat damage by gear.

3.2.4. Subsidies to ecosystem through the discarding of bait and by-catch

Potting for southern and western rock lobsters tends to be relatively benign in terms of damage through retention of by-catch and the fishing method itself would therefore be unlikely to impact the ecosystem through indiscriminate damage to the environment. In an ecological risk assessment undertaken on the western rock lobster fishery (Fletcher *et al*. 2005), a multitude of retained and non-retained species were identified as being taken in the course of fishing operations, but the overall the species by species affect on the environment was regarded by the assessment process as being low. Since the assessment was undertaken, action has been taken to mitigate the few moderate risks to the environment that were identified.

While the trophic effects of western rock lobster bait on the ecosystem has been considered to be relatively small, there has recently been published research suggesting that it could have positive effects on productivity of the fishery itself. The work of Waddington and Meeuwig (2009) showed bait to be a significant direct subsidy to the system, particularly to sub-legal lobsters. The mass balance model that they used in their study showed bait contributed a maximum of 13% of lobster food requirements over the whole ecosystem. It is hard to generalise on the applicability of this figure across the fishery because effort distribution is not uniform either spatially or temporally across the year. Other data (Waddington *et al*. 2008) showed the contribution of bait to western rock lobster food requirements as being as high as 35%, however Waddington and Meewig (2009) believe this figure is only likely in some months of the fishing season.

The potential effects on productivity through subsidy by bait bears consideration in terms of the contribution that it might be making to the resilience of the western rock lobster population. The bait used in the fishery is imported from other parts of Australia and even internationally and so is a true subsidy to the ecosystem. About 1.4 kg of bait was used for every kilogram of western rock lobster landed in the 1995/96 season (Jones and Gibson 1997) and this figure of ~14,000 t of bait being annually used by the fishery has continued to be assumed (Fletcher 2005). However, the recent large decreases in the amount of effort in the fishery would have had a substantial impact on bait usage. A key assumption in fisheries science is that the unexploited resource has a natural population size which is determined by the carrying capacity of the environment and that the sustainable yield from the fishery is surplus production. Subsidy by bait to the system, as has been shown by Waddington and Meeuwig (2009), would in effect be increasing the sustainable yield of the population, and possibly dampening some of the year-to-year changes in the environment. The direct and indirect ecological impacts of this bait subsidy are unknown, but likely to be significantly important for resilience of the ecosystem, given the likely level of impact on the rock lobsters. Given the likelihood of bait subsidy also to a range of the other species living in this ecosystem, it is unknown if the bait subsidy is likely to increase or decrease resilience of the populations of other species, or the ecosystem as a whole.

No information is available on the subsidy by bait to the southern rock lobster fishery in Australia, but according to Morgan (Chairman, Southern Rocklobster Limited, pers. comm.) fishers use around 3kg of bait for every lobster caught. This would equate to approximately 12,000 t of annual bait usage for catching southern rock lobsters in Australian waters. Given the higher bait usage per kilogram of southern rock lobster caught compared to western rock lobster, it would seem that subsidisation to lobster productivity with associated contributions to resilience may also be a significant factor associated with that fishery and the ecosystem where it operates.

An analysis of southern rock lobsters inside and outside a marine reserve in New Zealand (Freeman 2008) concluded that lobsters on the fishing grounds were found to be potentially deriving up to 46% of their tissue from fish species used as bait. The author considered that an increase in the natural mortality rate within the reserve through increased cannibalism and different foraging behaviours inside and outside the reserve were responses to increased competition for food in the reserve population. Once again, this highlights the contribution that protein inputs from fishing operations can have to the resilience of an exploited population and local ecosystems.

An inevitable consequence in every trawl fishery is the capture of unwanted species that are of no commercial value and that are discarded. In the SET fishery there are over 100 such species (AFMA 2009a). It is estimated that the retained catch for this fishery in 2006 was ~2800 t and that the discarded catch was ~10 t. There has been, and continues to be, a concerted effort to reduce the by-catch in this fishery which is very positive. It would seem unlikely that the current discarded catch would play a particularly significant role in terms of subsidising communities within the ecosystem. However, that may not have always been the case given that the fishery has been in existence for about a century and that in earlier years it would have had a different species mix, possibly fewer species with commercial value and that there was less emphasis then on minimising by-catch.

3.2.5. Habitat damage from fishing

Trawling has the potential to be damaging both directly to the fish stocks that they are targeting and indirectly to the ecosystems in which those fish populations occur. Not only do they remove the target species, but the nature of the fishing method means that other by-catch species are returned to the sea dead (see 3.2.4), or are processed as a by-product of the fishing operation. Repeated trawling can also prevent recolonisation of the grounds by slower growing benthic communities, and this in turn can modify the habitat and with that, the associated foodchains. The effects of fishing on the target populations are dealt with elsewhere (see 2.1.3; 2.2.3; 2.3.3.; 2.4.3. and 3.2.2.), but this section deals specifically with the indirect impact that trawling might have on the resilience of the case study populations of eastern gemfish and tiger flathead the south east trawl fishery.

Habitat in the south east trawl fishery has changed over time and this has not necessarily been a gradual process. Given that the fishery has been in existence for many years (Figs. 4 and 5), much of the softer ground has been trawled and retrawled many times over. However, in more recent years there have been technology developments that have allowed fishers to target previously unfished hard-ground habitat features that attract fish (Bax *et al*. 2000). Bax and Williams (2001) note that some fishers are now reporting substantial erosion and disappearance of seafloor features as a result of increased targeted effort.

Clearly any impact that affects habitat is going to impact the organisms colonising the associated benthic communities. Species which are fragile and long-lived are most vulnerable, with for example experimental trawling showing that some echinoderms and tube building polychaetes, may suffer reductions in density of up to 65% of their pre-trawling population numbers (Bergman 1992). In a meta-analysis of the responses of the different habitat types and associated biota to trawl gear, Kaiser *et al*. (2006) showed that in many instances recovery times can be measured in years, with slow growing sponges and soft corals taking up to eight years to recover.

As a result of a study that has mapped seabed habitat on the South-Eastern Australian continental shelf, Bax and Williams (2001) have been able to identify major seabed features in the area and assess their vulnerability to recover after modification. Since trawling is one of the potential modifiers of habitat, the research is particularly relevant to the management of the fished resources and the way that trawling might determinant the resilience of populations and communities associated with those habitats.

Gemfish are pelagic foragers, and their distribution is largely associated with the interaction between oceanographic and shelf-break features that produce conditions that are opportunistic for feeding and breeding (Prince 2001; Prince and Griffin 2001) and damage to benthic habitat is a less direct influencing factor than it would be likely to be for tiger flathead. Tiger flathead by comparison are benthic dwellers, which

Prince (2001) notes are protected by the limitations of trawlable grounds over part of their distribution (e.g. west of Bass Strait in depths of <200m). By implication this would mean that not only is part of the Tiger flathead broodstock protected in those untrawlable areas, but so too are their associated benthic communities.

Compared to trawling, potting for lobsters has a much lower likelihood of damaging habitat on the fishing grounds. A risk assessment of potential damage to habitat by fishing pots has been undertaken for the western rock lobster fishery rock lobster fishery and potential damage to different habitat types was considered to be minimal (Fletcher *et al*. 2005). The same outcome could be expected for pot fishing in the southern rock lobster fishery.

3.2.6. Effect of improvements in gear technology combined with range reductions from overfishing and the effect of hotspots on catch rates

As has already been noted, gear developments are allowing fishers to be able to trawl areas which were hitherto inaccessible due to the hardness and irregularity of the benthos (see Bax *et al*. 2000). The competitive nature of fishing means that fishers need to continually be developing knowledge about their target species and about how they can harvest them in the most effective and cost efficient way.

The East Coast trawl fishery has undergone many developments since its inception. Klaer (2001) has outlined some of the changes: in 1915 when the fishery began, steam trawlers were used and their numbers expanded until 1929 before decreasing, especially during World War II years. There was a resurgence of steam trawlers after the war, but after 1954 they were taken out of service with the last one leaving the grounds in 1961. Danish seine vessels began operating in 1933 with the fleet expanding before, as with the trawl fishery, declining over the war years and increasing thereafter to peak in the mid-1960s. Diesel trawlers entered the fishery in 1972 and their numbers peaked in 1991.

With these changes in vessel type and gear, came fisher ability to target different species and deeper depths. Also, experience led to better knowledge of the grounds and habitats and efficiencies of the gears that were being used (Klaer 2001). Colefax (1934 quoted by Klaer) records flathead catches which in the very early years of the fishery produced exceptional landings of very large breeding fish on the Botany Bay ground; by the 1930s those same grounds were practically useless.

The mid-1980 and early 1990s are considered to have been a particularly important era in terms of the development of improved fishing gear technology in the South East trawl fishery (Baelde 2001). This period saw the development of GPS and its later integration into plotter systems, a development which limits the time and skills needed to locate previously identified fishing grounds. From a positive stance, these electronic aids have assisted fishers in being able to reduce unwanted catch (Baelde 2001), which has been one of the challenges of the single species ITQ management system that was introduced in 1992.

The difficulty with the improvements in vessels, net technology, electronic equipment and accumulated fisher knowledge, is that it is extremely difficult to quantify and to take into account in the formal assessment of stocks. Robbins *et al*. (1998) for example, considered GPS plotters to have increased efficiency in the Australian Northern Prawn fishery by12%, but obviously these increases are not applicable across different trawl fisheries because of specific characteristics and goals of particular fishing operations, and so this percentage could be more or less in the SEF. The uncertainty as to whether a stock is recovering or not can be masked by doubt surrounding what part of its greater availability to the fishery is attributable to its apparent recovery, and what part may be attributable to increases in the efficiency of fishing.

A complexity that is related to increases in efficiency of fishing technology being able to pinpoint the target species, is that catch rates do not always respond to the abundance of the target species, in the way that might be expected. Some fisheries, for example the trawl fisheries for haddock on the southwestern Scotian shelf (Marshall and Frank 1995) and Atlantic cod in the Gulf of St Lawrence (Swain and Wade 1993), have recorded stocks contracting into optimal habitats at low population sizes, in what has been considered to be a response by fish moving to favourable habitat (or hotspots) from neighbouring areas of with less optimal habitat – the so called density-dependent habitat selection response of fish and other populations. The net result is that fishers are able to retain relatively constant catch rates in the face of a declining stock, because they are fishing optimal habitats which are being fed by the declining abundances of fish from the surrounding areas.

The effect that these hotspots can have on interpreting catch data, can lead to similar difficulties to those described in taking improvements in gear technology into account when using catch data as a proxy for stock abundance. Fish such as gemfish that aggregate along the shelf edge adjacent to topographic features such as canyons or bluffs (Prince and Griffin 2001) would be a classic example of a population that could have been impacted in the past as a result of their congregating behaviour, without catch rates indicating the state of the true state of their depletion. If this were the case, it would undoubtedly have had negative implications on the resilience of the population.

The effect of gear technology improvement and its effect on efficiency increases on catch rates in the western rock lobster fishery, has been an important focus of research in that fishery over the years (see for example Brown *et al.* 1995; Fernandez 1997). Because the fishery has been managed until the recent 2010/11 season under an effort control system, there has been a strong incentive for fishers to have the best available boats, electronic equipment, pots and bait, in order to optimise their share of the available catch. As a result, over the years boats became bigger and faster and were fitted with state of the art electronic equipment (GPS, echo sounders and associated equipment) as it became available.

Estimates of the increase in fishing power in the western rock lobster fishery between 1971/71 and 1992/93 were 0.5-2% per annum in shallow water and 1-4% per annum in the deep. Overall levels of efficiency increases for the fishery of 1-3% per annum have continued to be used, but they have been the subject of doubt in recent times, particularly by Industry themselves, who believe the annual increases to be considerably greater (Department of Fisheries, Western Australia 2008).

As noted earlier, the western rock lobster fishery has suffered a serious and as yet, unexplained downturn in puerulus settlement. One explanation for this low level of settlement from a risk assessment reported by Brown (2009) was that the brood stock might have fallen to below the 1980 threshold level – a scenario that was possible if efficiency increases were about 8.5% per annum, a figure which had been determined using a depletion analysis of monthly catch rates over one part of the fishing season.

The increasing ability of western rock lobster fishers to be able to accurately place their gear on optimal ground could be artificially inflating estimates of the stock size and egg production in the fishery. Catch rates in the fishery generally and egg production estimates which are based on mature female catch rates specifically (Melville-Smith *et al.* 2009), may be artificially inflated by relying on rates recorded on the optimal ground that is targeted by the fishers. Some 'old hands' in the fishery are happy to admit that the old method of haphazardly setting lines of pots that produced a living for fishers in earlier years, would produce little or nothing now. In the short term the effect of fishing 'hotspots' may be distorting the state of the resource and this may have an impact on resilience of the population in the long term.

These questions of undocumented increases in fishing efficiencies greatly complicate the interpretation of catch trends, particularly in the short term (say over 5 to 10 years). In the long term, these uncertainties become less important because eventually the catch data begins to better reflect the level of fishable stock after refuges and similar spatial distortions become resolved. Long term catch statistics (50 years or more) can therefore provide a more reasonable reflection of changes in population size and after stability is achieved, providing a better representation of short term fluctuations in response to the many factors that can act on species populations.

3.2.7. Genetic effects of fishing

Managers do not often consider the genetic consequences of exploitation in fished populations, but the need for fishing induced evolution (FIE) to be incorporated into management thinking is becoming increasingly evident (Allendorf *et al.* 2008; Jørgensen *et al.* 2008). The main reasons that evolutionary effects have been largely disregarded is that they are slow to be noticed and secondly, many of the more sophisticated genetic techniques that are now commonplace, were not available a decade or two back. The fact that other effects of fishing are far more obvious than fishing induced change has not helped promote the need for these effects to be considered.

Genetic change in a population as a result of fishing can be caused by selecting for particular heritable life-history traits, or by severely reducing the population size to low levels. Under very high exploitation rates it is possible for both of these effects to be operating on a fished population.

Selecting for particular life history traits such as size at maturity or growth rate, would be most likely to become evident in a fished population which has knife-edge selection at a given size (e.g. the legal minimum size), a short life history and high exploitation rates that force the breeding part of the population to be dependent on just a few year classes. Many invertebrate fisheries fall into this group, particularly as they often have good survival of individuals below the legal minimum size that are returned to the sea, a factor which makes any selection process particularly well defined.

Reducing a fished population to low levels and thereby impacting genetic diversity is in general most likely to be associated with populations having long life histories as these are usually the ones that are easiest to overexploit. In theory, genetic diversity should only be impacted when populations decline to very small numbers, but in practice the number of fish in a population can be very substantially larger than the numbers that are reproducing and that are therefore part of the genetic pool. In other words, even where a population size is large if only a small part of that population is effective in providing breeding success, genetic impacts may still be a dominant force in the population resilience expressed through the breeding component of the population.

There are numerous of examples of marine populations that have been impacted by fishing pressure leading to changes in heritable traits, for example the northern cod fishery (Olsen *et al*. 2004) and North Sea plaice (Grift *et al*. 2003). While there is no evidence that such change has occurred in the eastern gemfish, tiger flathead and southern rock lobster populations, although that possibility should not be excluded from consideration, there is evidence for a decrease size in maturity in western rock lobsters (already mentioned under climate change, see 3.1.1). It is very conceivable that this change in the size at maturity of western rock lobsters may have been caused by fishing-selective genetic change (Melville-Smith and de Lestang 2006; Allendorf *et al*. 2008), as well as a response to temperature changes; disentangling the possible cause of these type of observed changes in life history parameters is well known to be tricky (Kuparinen and Merilä 2007). The impact that a smaller size at maturity is likely to have on the resilience of the fishery – i.e., one of reduced productivity, has already been discussed (see 3.1.1.).

Loss of genetic variation in populations as a result of fishing has been recorded in a number of different studies, although as has already been mentioned, studies of this type are not yet commonplace. Examples of two Australasian studies that have shown these effects have been on the orange roughy (Smith *et al*. 1991) and snapper (Hauser *et al*. 2002) fisheries off New Zealand. No such studies have been done on tiger flathead, eastern gemfish or southern rock lobster, but an investigation is currently underway on the western rock lobster population.

The implications for the resilience of a population resulting from a reduction in genetic diversity are far-reaching. Loss of genetic variation can reduce the productivity of the exploited population both by reducing individual fitness in the short term and in the longer term by reducing the ability of populations to evolve (Allendorf *et al*. 2008). Some of the effects of climate change have already been discussed and it is clear that any reduction to the ability of a population to adapt to these changes will have potentially serious impacts on their resilience.

3.2.8. Selecting the big ones

Fishing generally targets large individuals and many fisheries have a legal minimum size to protect future recruits to the fishery. Often the legal minimum size is set in an arbitrary way at slightly above the size at maturity in order to give the females a chance to at least spawn once before capture. The targeting of large individuals in a population by fishing is the opposite of that occurring in the natural situation—large and mature animals in the prime of their life are the ones with the lowest mortality rates.

Although it has long been known that large females produce more eggs than smaller younger individuals, research is now frequently showing that eggs from small females do not have the same qualities as those from larger individuals. For example, in some fish species: eggs from old spawners have been shown to have a better hatching success than those from new spawners (Solemdal *et al*. 1995; Trippel 1998); older females have been shown to spawn longer and later into the breeding season (Hutchings and Myers 1993); eggs produced from older females have been shown to be larger and carry more oil (Morita *et al*. 1999) and this in turn has been shown to improve their survival rates (Berkeley *et al*. 2004).

Traditional management models take no consideration of the differences in quality of egg contributed by females of different age. However, where these differences have been incorporated into stock recruitment relationships, they have been found to have a very significant influence on the productivity of the stock (O'Brien *et al*. 2003; Field *et al*. 2008).

Information on whether there is any difference in the quality of eggs and larvae from different aged tiger flathead and eastern gemfish is not available and one can only speculate as to how the removal of older year classes from the spawning stock may have affected the resilience of these species.

Information is available however for western and southern rock lobsters, and size/age plays a very different role in the quality of eggs and larvae produced by the two species. In western rock lobsters, there is generally no significant difference in the quality of the eggs, based on lipid class, fatty acid composition, protein composition or water content for females of different size, age or area where they were captured in the fishery. Neither is there any difference in the competency of their larvae, based on their ability to survive long periods without food (Melville-Smith *et al*. 2007). In southern rock lobsters, the reverse is true: in that species, larger females have been found to carry larger eggs, with more lipid, that hatch larger larvae, that survive longer during periods of low food supply (MacDiarmid *et al*. 2000).

In addition, it has been shown that small male southern rock lobsters have less capacity to fertilise the big clutches of eggs that are produced by large females and furthermore, that the amount of ejaculate produced by male lobsters declines with their frequency of mating over a given period. If the ratio of large male lobsters to breeding females becomes severely distorted, the likelihood increases that egg production in the population will be negatively impacted by sperm limitation (MacDiarmid and Butler 1999).

What does this all mean for resilience in these two lobster species? In the western rock lobster fishery there is a management rule that protects both small and large females by having a minimum and maximum legal size, but no maximum size rule for male lobsters. The potential therefore exists for clutches of eggs, particularly of large females, to be inadequately fertilised in this fishery. However, on the positive side of egg production, the fact that there is no relationship in this species between the size of female and the quality of eggs produced, combined with the declining size at maturity that protects more of the brood stock, continuation of the current management arrangements should work towards maintaining the effective level of egg production.

The southern rock lobster fishery is expected to continue to be influenced by higher water temperatures which, as discussed earlier (see 3.1.1.), are expected to lead to larger lobsters on the more southerly Tasmanian grounds and perhaps lower abundances in the more northerly distribution of the species. From a positive point of view, some of the predicted decrease in southern rock lobster recruitment (Pecl *et al*. 2009) could be offset by quality and larger quantity of larvae likely to be produced by the predicted increase in the size of lobsters that will colonise the grounds, although this may be a temporary effect that becomes balanced by the effects of increasing temperature on recruitment.

3.3. Coastal zone development

Resilience of inshore marine populations may be strongly influenced by coastal zone development from a variety of different drivers. The temperate coasts, as in the rest of Australia are experiencing strong population growth trends leading to expanding economic, industrial and social pressure on coastal ecosystems. How the marine environment will withstand these pressures depends on how this development is managed.

Eastern gemfish have a relatively deep-water distribution and coastal zone development is therefore of limited relevance to considerations relating to this discussion. Tiger flathead have a much shallower distribution (10-400 m, most commonly shallower than 200 m (Kailola *et al*. 1993)), but even for them the impact of coastal development on their population would most likely be limited to an increase of recreational fishing pressure as a result of the expansion of coastal communities and perhaps some inshore development on the shallow fringes of their depth range.

Both western and southern rock lobster can be found in shallow water and coastal zone development would be relevant to those populations. In particular, the puerulus stage of western rock lobsters on which the whole population depends, settles on hard surfaces, in particular limestone reefs, from shallow sub-tidal depths down to 20 m where they shelter and feed amongst the naturally highly diverse and complex reefal systems (Fitzpatrick *et al*., 1989). Southern rock lobster pueruli settle deeper, down to at least 50 m (Booth *et al*. 1991), and would therefore be less impacted by coastal zone development.

Establishing the impact of coastal development on these populations is hard to gauge. The construction of harbours, wharfs, anchorages and similar structures could conceivably provide additional habitat for pueruli and juveniles. However, associated activities such as sediment plumes during construction, as well as noise, lighting, chemical runoff and shoreline changes as a result of longshore drift during and after construction, would be more likely to have a significant negative direct and indirect impact.

The Abrolhos Islands are considered to make a particularly important contribution to egg production in the western rock lobster fishery (Melville-Smith *et al*. 2009). During the fishing season permanent camps are occupied by fishers and the impacts of these camps and associated infrastructure were considered by an ecological risk assessment conducted on the fishery (Fletcher *et al*. 2005). Included in these considerations was nutrient enrichment from toilets, sinks and showers, disturbance and clearing of vegetation and the dumping of domestic waste into the ocean at the Islands resulting in a potential reduction in the ocean environment quality. Only the latter risk was given a moderate ranking, however since that time measures have been put in place for commercial and domestic waste to be brought back to the mainland for disposal.

3.4. Introduced marine pests

The introduction into Australian waters of several exotic marine species and their subsequent establishment is well documented (Wells *et al*. 2009; Anon. 2011). Probably one of the best known examples is that of the establishment of pacific oysters on the East Coast of Australia and the dominance of this introduced species over the endemic east coast Sydney rock oyster. The concern with these type of introductions is that they have the capacity to outcompete native species for food and habitat, and relegate the native species into minor ecological niches and functions, and threaten their resilience and ultimately their existence.

While there may be more, only one introduced pest has been recorded as potentially playing any substantial negative influence in the ecosystems of the populations that form the case studies in this report. Williams *et al*. (2000), in their intensive ecosystem survey of south-eastern continental shelf, recorded the New Zealand screw shell (*Maoricolpus roseus*) as a major component of the fauna that they sampled in depths of 25-80 m.

They believe the screw shell to have been introduced into Australia from New Zealand in the 1940s, brought in attached to oysters that were imported from that country at the time. Numbers of the screw shell have grown rapidly in the fifty years since its introduction, as has its distribution. According to Williams *et al*. (2000), over that time it has moved from the Derwent River, up the east coast of Tasmania, across Bass Strait in the 1980s and at the time of their survey in the late 1990s, it had been found in Sydney Harbour. They recorded some areas along the coast with densities in excess of 1000 individuals per square metre.

Williams *et al*. (2000) suggest that the screw shell where it has become abundant has most likely had a substantial impact on the ecosystem. In these areas it has displaced native gastropods and has provided shelters for hermit crabs leading to an increase in their numbers. They consider that because of their solid shells, screw shells are less available to some of the fish predators than shellfish that previously occupied that habitat and they speculate that the net result could be a severe reduction in the productivity of some fish species.

Australian authorities are well aware of the potential consequences that introduced pests can have on valuable fisheries and the potential for introducing exotic species is increasingly being considered in the management plans supporting new marine infrastructure (e.g. Oakajee Port and Rail 2010).

3.5. Pathogens

Infectious diseases are a known cause for population declines, even species extinctions and so clearly this is potentially an important factor when considering resilience of populations.

The Australian marine management community has become very aware of the disastrous effect that pathogens can have on marine fish and invertebrate populations, as a result of the recent pilchard and abalone viruses. In both cases, the viruses spread across populations on the south coast of the country. In the case of pilchards, the population was infected by a pilchard herpes virus (PVH) and about 60% of the stock was lost (Whittington *et al*. 2008). The abalone virus outbreak was more recent. Known as abalone viral ganglioneuritis (AVG), it was first recorded in the marine environment in Victoria in 2006 and has infected stocks along the coastline from its point of origin where it was first identified (Anon. 2008). The virus is highly virulent and where reefs have been infected by AVG, abalone losses of up to 95% have been recorded (Anon. 2008).

The monitoring of infectious diseases in marine wild populations, particularly of invertebrates, is not routine and it is likely that many outbreaks go unreported. Outbreaks of infectious diseases that have been reported have been related to a variety of anthropogenic disturbances, ranging from eutrophication, overfishing (Jackson *et al*. 2001) and aquaculture. In recent times, the warming effects of climate change have become a consideration in terms of disease risk; in their review, Harvell *et al.* (2002) predict that in most instances the frequency and severity of disease transmission will increase as a result of climate warming leading to increased pathogen development and survival rates, disease transmission and host susceptibility.

Predicted increases in water temperature along the Australian seaboard have already been discussed (see at 3.1.1) and forecasts are that inshore water temperatures will increase by around 1^{o}C by 2030 and 2-3^{o}C towards the end of this century. Warmer conditions result in most pathogens responding by increasing their growth rate and range extension). Harvell *et al*. (2002) suggest furthermore, that hosts once attacked by a pathogen can become targets of opportunistic infections by other pathogens and if are under stress from elevated water temperatures, there may be little capacity for the hosts to recover from infection.

There have been no records of virulent diseases in the four species discussed in this case study, but clearly that potential does exist. Shields *et al*. (2006) for example have no doubt that there will be more pathogenic viruses discovered such as the one occurring naturally in juvenile lobsters in the Florida Keys, as diagnostic tools develop and as more monitoring programmes are put in place. The impacts that high mortalities resulting from a virus outbreak could have on our lobster populations are obvious.

Western and southern rock lobsters are prone to bacterial infection of the shell. This type of infection is common in all species of lobster and can occur as a result of injury or exposure to pollutants (Shields *et al*. 2006). Shell disease is generally more common in older animals because they moult less frequently than juveniles and usually the condition is overcome when the animal moults (Shields *et al*. 2006).

Damage from fishing is a common cause for the introduction of shell disease in the discarded (undersized) catch. Freeman and MacDiarmid (2009) for example recorded southern rock lobster in fished areas having high rates of bacterial infection of the tail (known as tail necrosis) compared to unfished areas. Although these physical affects from fishing deserve consideration, they are preventable to some extent through escape gaps for undersized lobsters and good handling practices which are features of western and southern rock lobster industry code of practice manuals (Stevens 2004a;b) and would seem unlikely by themselves, to affect resilience.

One of the most important factors affecting the health of lobsters is considered to be stress, which is a normal physiological response to changes in environmental conditions (Evans 1999). These conditions can include changes in water quality parameters (e.g. temperature, pH and oxygen levels), physical factors (e.g. handling), behavioural interactions, or nutrient availability. Future changes in the marine environment in terms of increased temperature and ocean acidification, have already been discussed (see 3.1.1 and 4.1.4) and the possibility of increased threats to marine populations from disease as a result of the stressors from climate change, should not be disregarded.

3.6. Governance and socio economic

The objectives and the planning that are upheld by governance arrangements are important determinants of the resilience of a fished population to exploitation. Some of the history surrounding the management over the lengthy periods of development of the fisheries for the four case study species has already been outlined (see 2.1.2; 2.2.2; 2.3.2 and 2.4.2). The object in this section is to briefly discuss the way that the species have been managed and in particular, the way that governance of those fisheries has responded to socio economic considerations, and how that in turn has impacted resilience of the exploited populations.

3.6.1. Tiger flathead and Eastern gemfish

As noted earlier, eastern gemfish and tiger flathead are two of (currently) 34 species managed by the AFMA under the umbrella of the SESSF. Since these two species are managed under the same management plan (the Southern and Eastern Scalefish and Shark Fishery Management Plan 2003), they are dealt with together under this heading of socio-economic governance.

According to AFMA (2009b): The SESSF is currently managed by output controls. TAC limits are allocated as individual transferable quotas (ITQs) or quota statutory fishing rights (SFRs) which are fully tradeable and can be permanently transferred or leased. Around 80% of the total commercial landed catch comprises quota species, with 34 species/species groups (of which eastern gemfish and tiger flathead are two) being managed through quota allocated as ITQs or SFRs. There are controls on gear types used, net mesh specifications, trip limits, incidental catch limits, size limits for some species, and prohibited take of other species. Input controls used include limited entry; gear restrictions; and spatial and temporal closures. In addition, industry implements voluntary measures in co-operation with AFMA, including industry Codes of Conduct. Management measures used in conjunction with the Management Plan are continuously reviewed. The fishery has been granted Wildlife Trade Operation (WTO) approval under the EPBC Act, acknowledging that it is managed in an ecologically sustainable manner, since 2003.

Consultation with the fishery is through the South East Trawl Management Advisory Committee (SETMAC) and in the case of both eastern gemfish and tiger flathead, the Shelf Resource Advisory Groups (ShelfRAG) provides advice on the status of the stocks and on the impact of fishing on the environment. A formal risk assessment of the effects arising from the trawl fishery has been undertaken by Wayte *et al*. (2007) and management of the stocks and their possible impact on the ecosystem is guided by this assessment.

The SESSF has harvest strategies with associated biological reference points, to meet particular objectives for the stocks being managed. According to Anon. (2010), both the eastern gemfish and tiger flathead stocks will have (2011/12 season) reference points of 20:40:40 (B_{lim}; B_{msy}; F_{tar}) to the point that fishing mortality reaches F_{48}. Once this point is reached fishing mortality will be set at F_{48}, because Day (2008, quoted by Anon. 2010) has determined for most SESSF stocks where the proxy values of B_{40} and B_{48} are used for B_{msy} and B_{mey}, this form of rule is equivalent to a 20:35:48 strategy. These rules are set to provide for a low risk of overfishing, and although they also protect against major impacts of fishing, they take into account only minimally the trophic impacts and detailed needs of ecosystem function where the fishery operates. As a result, the governance decisions applied to these fisheries have an uncertain impact on resilience of the fished populations and their functional role in their ecosystems.

The eastern gemfish is one of three species in the SESSF that is considered to be overfished. Under the Commonwealth Harvest Strategy Policy 2007, there is a requirement for any stock with a biomass below the limit reference point to be placed under a stock rebuilding strategy and such a strategy has been developed for eastern gemfish in consultation with the Department of Sustainability, Environment, Water, Population and Communities (SEWPaC) and other stakeholders.

As with any multi species fishery, the SESSF has issues with discard rates. One of the responses to this has been a program introduced by AFMA to specifically deal with bycatch and discarding in commonwealth fisheries AFMA (2009a).

Compliance, another integral part of fisheries governance, uses a multi-targeted approach of (i) compulsory vessel monitoring systems on all vessels; (ii) education through the simplification of regulatory requirements associated with management plans; (iii) the undertaking of routine and targeted compliance activities, both at sea and ashore; (iv) aerial surveillance to assist on-ground activities; (v) intelligence reporting and (vi) information programs though liaising with external stakeholders and an enforcement hotline (CRIMFISH) (AFMA 2009b).

Governance in the SESSF is apparently much more thorough now than it was in the last millennium when scientific recommendations were not routinely accepted, with resulting negative implications to some of the stocks (see Tilzey and Rowling 2001; Bax *et al*. 2005). Eastern gemfish is probably one of the stocks that was a victim of such outcomes and it could be argued that the apparent lack of resilience in that population today may have its roots in the governance of the fishery during the late 1980s through the 1990s management era.

The economics of the fishery is much improved to what it was a few years back. Prior to 2005, several Commonwealth fisheries were experiencing unsustainable fishing effort and declining profitability. The Commonwealth Government addressed this in 2006 by implementing a structural adjustment scheme aimed at reducing effort and improving the profitability of those who chose to remain in the fishery (Viera *et al*. 2010). The fishery buyback component of the adjustment package to be used for the purchase of concessions in target fisheries, one of which was the Commonwealth trawl sector of the SESSF, was $149 million (Viera *et al*. 2010).

The results of the package have been assessed by Viera *et al*. (2010) by comparing statistics in the fishery in 2005/06 before the buyback, with 2007/08 after the buyback. Over that period, boat levels in the Commonwealth trawl sector were reduced by 40% (81 to 49 boats), catch per boat increased by 26%, real revenue per boat by 44% and costs per boat by 24%. The net result has been a very substantial improvement in economic returns for those remaining in the fishery.

In February 2009, Eastern Gemfish were placed on the national threatened species list as 'conservation dependent', but stakeholders are not unanimous that all that can be done to manage threatened species, is being done. For example in a submission (WWF 2008), WWF-Australia stated their opinion that many opportunities to protect and manage species have been lost through a government mindset that has appeared to often seek to bend the requirements of the Environment Protection and Biodiversity Conservation Act in order to facilitate a continuation of a business-as-usual-approach, particularly in areas such as Commonwealth fisheries.

3.6.2. Western rock lobster

The western rock lobster fishery was the first fishery in the world to be certified by the Marine Stewardship Council (MSC) as a well-managed fishery. To be awarded this third party certification, the fishery had to satisfy three overarching principals against which it is assessed: those of sustainability, minimizing environmental impact and effective management. Effective management considers all aspects of governance and these have been assessed since 2000 when accreditation was awarded to this fishery, as being satisfactory subject to the completion of a number of conditions for the maintenance of certification.

The fishery has until 2010 been operating under input controls, but as of the 2010/11 season has moved to output controls, although still with many of the management measures that were in place when it operated under effort controls, for example: limits to the number of pots that can be used, restrictions on the dimensions of the pots and spatial and temporal management restrictions.

The fishery has been utilizing a limit reference point for egg production based on levels measured in 1980. This reference point was based on historically low levels recorded in the fishery and the object has been to avoid those levels being recorded again in the future (Fletcher *et al.* 2005). The fishery is now moving to an updated and improved set of decision rules (Donohue 2010) in line with the Commonwealth Government's Fisheries Harvest Strategy Policy. In brief, the new harvest strategy will have maintenance of egg production above the 1980s level as a target and will have egg production 20 per cent below that level as a limit reference point. A maximum economic yield harvest rate (i.e. the level of exploitation that provides the greatest economic benefit) will form the target harvest rate value.

There is formal recognition of recreational fishing and the 2009/10 season was the first year that formal catch allocations of 95% to the commercial sector, 5% to the recreational sector under the Integrated Fisheries Management process.

A great deal of effort is applied to keeping high standards of compliance in the fishery and the industry is at the forefront of this field (McKinlay and Millington 1999). Approximately 2-2.5% of the commercial catch is inspected by Fisheries and Marine Officers at offloading and numbers of non-compliant animals in the catch are 1-1.5 per 1000 checked (Fletcher and Santoro 2010).

Perhaps the least commendable aspect of governance in this fishery is the inadequacy of its consultation processes for providing stakeholders the opportunity to express views and concerns associated with the management of the fishery. This aspect was identified as a weakness of the fishery in its first MSC certification assessment in 1999 (SCS 1999) and it appears to have deteriorated if recent comments by stakeholders (e.g. Raphael 2010) and the MSC Surveillance Audit (SCS 2011) are anything to go by. The impact on the resilience of the fishery and its associated ecosystem of having effectively excluded NGOs from having any significant input into the management process, can only be speculated on.

The integral role that stakeholder involvement plays in management and governance has been well illustrated through qualitative modelling of the role of direct stakeholders in influencing government decisions (Metcalf et al. 2009). The stability of their model was substantially greater when stakeholder cooperation in management was included.

From a socio economic point of view, management measures introduced as a result of low levels of settlement have resulted in severe reductions in landings in the fishery in recent years. This combined with low beach prices, increased labour costs, high fuel prices and general uncertainty in the fishery has seen the number of boats in the fishery halve over the last five years. Obviously this has had severe impacts on coastal communities and activities associated with the lobster fishery, such as boat building, maintenance and catch processing.

Several different social and economic studies have been undertaken on the western rock lobster fishery over the years. However, now that the fishery has moved to output controls and that the fleet size and tonnage landed have decreased so drastically, that research has less relevance and is outdated.

An investigation of the social issues in the fishery undertaken by Huddleston (2006) at a time that changes in management arrangements were being considered (i.e. 2002 to 2005) showed that the resilience of communities to change is very variable across the cities and towns that are associated with western rock lobster fishing activities. In general, towns in the centre of the fishery (from Mandurah to Lancelin) were more resilient, while those south to Augusta and north to Kalbarri were more fragile and sensitive to change, related to likely patterns of change in catch and home bases for the fishing fleet.

3.6.3. Southern rock lobster fishery

The governance arrangements and socio-economic outcomes in relation to southern rock lobster resilience for the commercial fisheries across four states, and within three of those states spanning more than one management zone, is overly complex and beyond the scope of this report.

However, in general terms, all states have regulations enforcing a legal minimum size and the return of egg-bearing animals to the sea. Three states (South Australia, Tasmania and Victoria) are members of Southern Rock Lobster Ltd., an independent company owned by licence holders. The fishery has a pot-to-plate accreditation program known as Clean Green which is owned and maintained by Southern Rocklobster Ltd (Southern Rocklobster Ltd 2008a) and much of the product is marketed using that trademark.

With the exception of Western Australia, all states with commercial southern rock lobster fisheries now operate under quota arrangements. South Australia first introduced quota in the southern zone of the fishery in 1993/94, followed by Tasmania in 1998, Victoria in 2001 and the Northern Zone of South Australia in the 2003/04 season (Gardner 2008).

Gardner’s (2008) observation on profit maximization in the Australian southern rock lobster fisheries has been that this has resulted in very vague management goals and that as a result, management performance has been sub-optimal. The cited examples being Victoria and the Northern Zone in South Australia where catch (quota) was kept high in the false belief that it would help struggling fishers, but in effect made their situation worse. The Northern Zone of the Southern Rock lobster fishery has had a particularly catastrophic recent catch history, falling from 1001 t in the 1999/2000 season to 503 t when a TAC was introduced in 2003/04 (Linanne *et al*. 2010) and thereafter failing to catch the ever declining TAC until the 2009/10 season.

The decline over several years in southern rock lobster stock status across southern-eastern Australia has been considered by Linanne et al. (2010) to be due to a combination of management decisions and possible large-scale environmental influences. The same authors make the point that this situation means that there is probably a reduced number of females contributing to larval production and that there is therefore a need for conservative TACs to be set across the south-eastern part of the fishery for some considerable time into the future to protect existing biomass. This recommendation would seem to be an important and relevant one in terms of potentially protecting resilience of the stock over shorter-term socio-economic considerations.

Socio-economic considerations in this fishery should not be underestimated. According to Southern Rock Lobster Limited (2008b) the fishery is valued at nearly AUD$200 million and generates approximately 1600 jobs in South Australia, 1400 in Tasmania and 400 in Victoria, injecting almost $1/2 billion into regional economies annually. Obviously the impact of the downturns in the fishery over the last few years have been of concern to state governments, and the Victorian Government for example, undertook an AUD$5 million buyback of licences and quota in 2009 under a restructure aimed at improving the economic viability of the industry (Hillard 2009).

3.7 How factors may be interacting

In this review it has been necessary to deal with the different factors that can affect population resilience singularly, but there is considerable potential for factors to interact with each other.

The environmental effects of climate change for example, are very tightly linked with each other and so changes in wind patterns will affect circulation patterns, which in turn will affect sea temperature, all of which could be acting on the resilience of populations. Similarly, it is possible for several different effects of fishing to be influencing population resilience in unison, for example an intensively exploited fish population may be impacted by recruitment overfishing and at the same time have its resilience impacted by habitat destruction.

The examples given of how factors can interact have used the broad headings of climate change and fishing, but factors can also interact across far less closely related factors. Poor governance can be giving rise to overfishing, which in turn can be leading to recruitment overfishing and perhaps even loss of genetic variation in the population. A depleted population in its genetically compromised state, may be less capable of adapting to changing climatic conditions or the introduction of a harmful pathogen, leading to an every downward spiral of events and ultimately, perhaps even an inability to recover.

4.1 Conclusions

Marine animals are innately resilient. Their life history patterns often depend on larval phases that can be disrupted by changes in weather patterns and ocean currents, often with extreme consequences to recruitment levels. For example, collectors that are used to monitor the settlement levels in the western rock lobster fishery can have over 20 times more pueruli settling on them in one year than another (Phillips *et al.* 2003) but this does not translate into such large fluctuations in catches. When pueruli are abundant their mortality is high and this effectively buffers the number of lobsters within a cohort that survive to become adults. The result is that the maximum annual commercial western rock lobster landing over the last 30 years has only been approximately double that of the minimum annual landing over the same period, rather than the 20-fold difference in puerulus abundance measured by the collectors (Phillips *et al.* 2003). Similar patterns also operate at the ecosystem level, generally considered as processes contributing to the term 'natural mortality', except they are far more complex because of cycles in the abundance levels of different populations within the ecosystem that may interact with each other.

Stakeholders of all descriptions frequently hold a view that populations and their sustainable harvests, or fisheries production, should remain within reasonable bounds over time. After all, socio-economic considerations rely on a degree of stability from year-to-year to keep fishing fleets and supporting industries in business. Resilience as an ecological concept, does not automatically validate this expectation, and indeed vastly complicates the establishment and maintenance of stability and predictability in commerce. It has already been noted in Steven Cork's (2010) article that resilient ecosystems might have different proportions of their constituent populations while still retaining the essential function and identity of their ecosystem. It is the ability of the ecosystem to retain its identity even though the proportion and composition of its constituent populations may change over time that dictates its resilience. In some cases this may mean dominant species that lack resilience being replaced by one or more functionally similar species.

The question of what attributes characterise resilient populations and ecosystems has been widely considered by authors (e.g. Hutchings and Reynolds 2004; Palumbi *et al.* 2008; Suryan *et al.* 2009). In a meta-analysis of 90 fish populations, Hutchings (2001b) compared the response of the populations 5, 10 and 15 years after their collapse. The vast majority of populations continued to decline after their collapse, whether or not fishing mortality decreased. Furthermore, the extent to which fishing mortality decreased after the population had decreased, had no impact on their recovery up to 15 years later. The implication of this is that while fishing mortality must play a part in influencing resilience, there are other forces acting on a population and the ecosystem of which it is a part that determine its ability to recover. This confirms that notion that while one form of pressure may act to shift a population to a lower level and perhaps different ecological functional space in an ecosystem, once at that level it can adopt a new form of resilience that prevents its recovery to the former situation. In this sense, the resilience of the population then acts against recovery.

Jackson *et al.* (2001) have put forward an explanation of how these other forces might be acting to prevent recovery. They used a variety of datasets, in some cases going back as far as paleoecological records, to support their hypothesis that overfishing precedes other anthropogenic disturbances that lead to collapses of coastal ecosystems. They maintain that events that follow overfishing often include pollution, eutrophication, physical destruction of habitats, the introduction of pest species and in recent times, climate change effects. In their view, systems that are compromised by the effects of overfishing are made more vulnerable to these additional disturbances opening up the way to their potential collapse. Increasingly, evidence is pointing to populations being able to within limits, be resilient to short-term fluctuations caused by environmental or anthropogenic stress. Hammer *et al.* (2010) reviewed a range of factors that have been associated with the recovery processes of depleted commercially fished stocks. A key factor contributing to the recovery of collapsed stocks, was how rapidly fishing mortality was reduced after the collapse. In most cases, a lack of recovery was identified with fishing pressure being reduced 'too little, too late'. The critical element is the population or system not be moved to the point where it is replaced by an identity with different functions, structures and feedbacks.

The factors affecting resilience have been discussed at length in this report. However, there are some characteristics that are common to populations that have proved to be resilient that have not been mentioned in any detail.

Palumbi *et al.* (2008) note that recovery of ecosystems can be hindered by complex and often indirect species interactions. One of the factors which help to make ecosystems more resilient to change is when ecological redundancy (i.e. species that perform similar functions) is high, because this allows other species to potentially replace one or more key species in the ecosystem (Palumbi *et al.* 2008). Species-rich biotas are more likely to have greater levels of functional redundancy and this can assist in affording them a degree of ecological insurance against uncertainty (Hughes *et al.* 2005). Populations in highly diverse ecosystems are therefore more likely to be resilient to change – an observation borne out to some degree by Palumbi *et al.* (2008) who note that a smaller fraction of commercially fished species have collapsed and there has been a higher rate of recovery of collapsed species in diverse ecosystems compared to systems that are naturally low in species numbers.

Life history characteristics of a population is another trait that plays an important role in determining its resilience to one or more forces. Organisms that are fast growing with an early age at maturity and therefore a short generation time and that produce many offspring (termed r-selected), have the ability to survive in unstable environments because of their capacity to 'bounce back' after a population 'crash'. The opposite to r-selected organisms are slow growing animals that produce fewer offspring, typically found in more stable environments. These animals are referred to as k-strategists. The fast reproducing r-strategists are the first to recolonise a system after it has been destabilised, but over time with stability, systems tend to move towards colonisation by greater proportions of long-lived k-strategists.

The four fished populations that have been used as case studies in this report, provide a useful contrast to each other in terms of their resilience. Eastern gemfish would appear to be the least resilient of the four, with no obvious recovery to the population since the late 1980s when numbers fell so dramatically. With little or no response to management measures aimed at rebuilding the stock, the question arises whether the current status of the population is related to changes in the way that the ecosystem is functioning, a lack of ability of the population to adapt, or a combination of these issues. Tiger flathead by comparison, has proved over years of exploitation, to be remarkably resilient.

The two lobster populations have been resilient to date, but both are currently going through a downturn across their distributional range. In western rock lobster, there is uncertainty as to whether the downturn is related to environmental effects of brood stock issues; in southern rock lobster there is more confidence that the downturn is related to larval recruitment to the coast having been directly affected by environmental conditions in recent years (Linanne *et al.* 2010). In both cases, future landings are unlikely to be as large as historical landings because harvest strategies in these fisheries are moving towards optimising profitability rather than maximising yields (Phillips *et al*. 2010).

Despite the contrast in terms of their apparent resilience, the four species that form the basis of this case study have some similarities in their life history patterns. All are seasonal spawners that take several years to reach maturity. The two lobster species take a year or two longer to reach maturity compared to the two fish species, but they also have substantially longer life-spans than the fish. There are degrees of r and k-strategy and while the populations in this case study would be termed r strategists, they do not have the short generational turnover times that characterise perhaps more resilient species such as clupeid fish and penaeid prawns.

4.1. How can we prepare for anticipated pressures

It is clear that there are many factors that can potentially impact population resilience. While it is tempting to attempt to identify a single factor, it is likely that in most cases where populations fail to recover from a 'shock', that their lack of response is probably attributable to a combination of factors and associated considerations.

Few ecosystems "bounce back" to the same state they were in before a shock. It should therefore be no surprise to see populations wax and wane over time due to impacts on the ecosystem from drivers of change. It was noted in Steven Cork's (2010) contribution, that humans tend to focus on rapid change and are slow to appreciate less obvious, but not necessarily less relevant, change. The frog in boiling water anecdote springs to mind, where the frog will jump out if exposed to boiling water, but is comfortable and does not attempt to escape in water that is gradually warmed to the point that the frog is cooked. Scientists, managers and even the general public are quite rightly, quick to identify and attempt to curb overfishing or damage due to irresponsible fishing practices, but they have been slow to respond to less obvious signals such as those due to climate change (see 3.1) and fishing induced evolution (see 3.2.7).

We live in a particularly dynamic period of social, economic and environmental change. There is little doubt that the effects of climate change, expanding coastal populations and demanding socio-economic expectations will test managers of marine resources in the future. Their challenge will need to be one of acting in a precautionary manner to constrain potential environmental stressors so as to minimise those factors that could impact population resilience.

A positive step in this regard in fishing has been the widespread embracing of ecosystem-based fishery management principles. Also, in the case of exploited stocks, a move away from management strategies that aim to achieve maximum sustainable yield, to harvest rules with lower exploitation rates targeting maximum economic yield.

A second positive step would be to focus more research effort at predictive science so that action can be taken before events. Western rock lobster research has been in the forefront of this type of application (Phillips *et al*. 2007) and short to medium term management is very reliant on these abilities (e.g. de Lestang and Caputi 2011).

Finally, managers need to pay serious consideration to retaining diversity – be it biodiversity at the ecosystem level, or genetic diversity at the population level. Both forms of diversity have been identified as playing a crucial role in promoting resilience; biodiversity by assisting undesirable change from occurring and genetic diversity by providing the ability for organisms to respond to change where it has occurred. This latter ability will be particularly important in future years as the effects of global warming play an ever-greater role in shaping our environment.

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