



Complex interactions in a rapidly changing world: responses of rocky shore communities to recent climate change

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ABSTRACT: Warming of the planet has accelerated in recent years and is predicted to continue over the next 50 to 100 yr. Evidence of responses to present warming in marine ecosystems include shifts in the geographic range of species as well as in the composition of pelagic and demersal fish, benthic and intertidal assemblages. Here we provide a review of the changes in geographic distributions and population abundance of species detected on rocky shores of the NE Atlantic over the last 60 yr. This period encompassed the warm 1950s, a colder period between 1963 and the late 1980s and the recent period of accelerating warming to levels above those of the 1950s. The likely consequences of these responses are then explored. To do this, a summary of the dynamic balance between grazers, macroalgae and barnacles in structuring mid-shore communities is given before outlining experimental work on interactions between key components of rocky shore communities. Modelling and quantitative forecasting were used to predict changes in community composition and dynamics in a warmer world and their consequences for ecosystem functioning discussed. We then identify areas that need further work before making a case for the use of rocky shore species not just as inexpensive indicators of change offshore, but as tractable models to explore the direct and indirect effects of climate change in marine and coastal ecosystems. We also provide a societal perspective emphasising the value of long-term studies in informing adaptation to climate change.

KEY WORDS: Climate change · Rocky shores · Time series · Grazer-algae interactions · Ecological forecasting · Adaptational policy · Europe

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1. INTRODUCTION

Warming of the planet has accelerated in the last few decades (IPCC 2007). Some of the most rapid changes are occurring in the NE Atlantic (IPCC 2007), where increases of up to 1°C have been recorded in areas such as the Western English Channel (Hawkins et al. 2003). Marine ecosystems have responded to this warming with changes in abundance and shifts in geographic ranges of plankton (Beaugrand & Reid 2003, Beau-

grand & Ibanez 2004, Hays et al. 2005), fish (Brander et al. 2003, Beare et al. 2004, Genner et al. 2004), and off-shore benthic organisms (Hiscock et al. 2004). Some of the most marked and best-documented changes have been seen on rocky shores in Europe (Herbert et al. 2003, Mieszkowska et al. 2005, Simkanin et al. 2005, Lima et al. 2006, Mieszkowska et al. 2006, Herbert et al. 2007, Lima et al. 2007a), the United States (Barry et al. 1995, Harris et al. 1998, Sagarin et al. 1999, Zacherl et al. 2003, Harley et al. 2006) and South America (Ri-

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vadeneira & Fernandez 2005). In general, coldwater adapted species are decreasing in abundance and retreating polewards while warm water species are increasing in abundance and advancing.

Phenological changes have also been recorded in the NE Atlantic in relation to past climatic fluctuations (Sims et al. 2001, Sims et al. 2004) and phase shifts have been shown as a result of recent change (Edwards & Richardson 2004). These phase shifts may have major consequences throughout food webs including influencing the recruitment of commercially important species (Beaugrand et al. 2003, Edwards & Richardson 2004).

Long-term data sets are particularly valuable in putting recent changes into context. Current abundances and biogeographic distributions of species can be compared to those of previous warm periods such as the end of the 19th century and the middle of the 20th century (Fig. 1). Some of the most comprehensive data sets in terms of species recorded and temporal extent, albeit interrupted, have been collected in the Western English Channel (see Southward 1980, Southward et al. 1995, Hawkins et al. 2003, Southward et al. 2005) where links between climate and ecosystem fluctuation have long been recognised (Southward 1963, Russell et al. 1971, Russell 1973). Charting changes in species abundances, range and assemblage composition is a first step in understanding the influences of climate on community structuring and dynamics and the consequences for ecosystem functioning. While many changes in spatial and temporal patterns have been described, much less work has been done on the consequences for community and ecosystem processes, partly because of the experimental intractability of open offshore ecosystems. Much, however, can be learned from rocky shores, drawing on both the rich heritage of field-based experimentation (Connell

1961a,b, Paine 1966, 1979, Raffaelli & Hawkins 1999, Bertness 2007) as well as extensive long-term data sets and broadscale baselines, particularly in Europe (Fischer-Piette 1936, 1955, Crisp & Southward, 1958). Rocky shores are relatively simple ecosystems; the ecologies of many species are well known; this makes them an excellent model system for understanding the consequences of climate change for community and ecosystem processes.

In this review, stemming from an invitation to give a keynote presentation at the 2007 European Marine Biology Symposium in Kiel, we focus on European rocky shores where extensive long-term datasets spanning cooler and warmer time periods exist. The NE Atlantic region is also experiencing rapid temperature increases above global averages, in response to the recent period of climatic warming. In addition, there is little exploitation (artisanal or commercial) of rocky shore species, particularly in northern Europe, which could make detection of climate-related assemblage responses difficult. We give a brief overview of changes detected over the last 60 yr encompassing the warm 1950s, the colder period between 1963 and the late 1980s and the recent period of accelerating temperature increase (these changes are discussed in more detail in Southward et al. 2005). The likely consequences of accelerating climate warming are then explored at community and ecosystem levels. A summary of changes in mid-shore assemblage composition with latitude is given before outlining experimental work on interactions between key components of the community which are known to be responding to climate change. Modelling and quantitative forecasting are then used to predict community composition and dynamics in a warmer world and what the consequences will be for community structure and ecosystem functioning. A case is made for the use of rocky shore species as inexpensive indicators of change offshore. Rocky shores also provide a tractable model ecosystem in which to explore the likely direct and indirect effects of climate change on marine assemblages. We conclude by providing a societal perspective emphasising the importance of long-term studies in informing adaptational policies.

2. CHANGES IN ABUNDANCE AND DISTRIBUTION OF ROCKY SHORE SPECIES ON EUROPEAN SHORES

Changes in abundance and range limits in response to climatic fluctuations have long been recognised on rocky shores. Yonge (1949) and Wilson (1935) in their classic books on the seashore, report such fluctuations including the appearance and disappearance of the

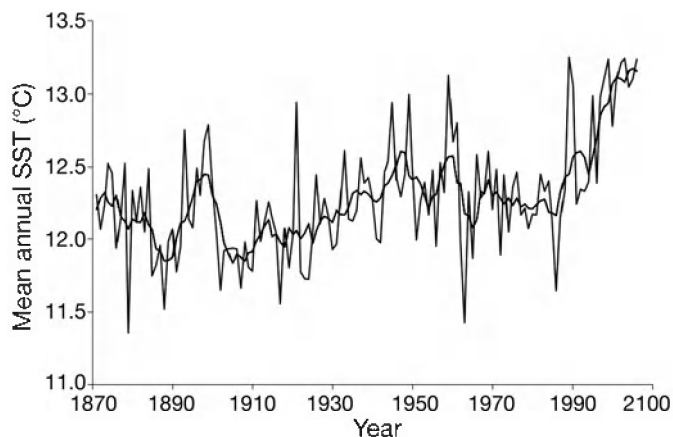


Fig. 1. Mean annual sea surface temperature (SST) for 1871–2006 off Plymouth (grid square 50–51° N, 04–05° W). Data from the UK Meteorological Office Hadley Centre

boreal arctic limpet *Tectura testudinalis* at its southern limit on the northeast coast of England. Southward & Crisp (1954b) observed that the northern species of barnacles *Semibalanus balanoides* was much rarer in the 1950s than in the 1930s in the English Channel. They suggested that this was caused by climate mediated competition with the southern species *Chthamalus stellatus*, which was becoming more abundant. This prompted the initiation of long-term observations which showed that in the southwest of England, the relative proportions of northern and southern barnacles fluctuated with temperature with a 2 yr time lag. *C. stellatus* became rarer in the cooler 1960s (Southward 1967) before increasing in abundance when conditions began to warm in the late 1980s (Southward 1991, Southward et al. 1995, Southward et al. 2005; Fig. 2). These time series were largely interrupted by a lack of funding, but when restarted in the late 1990s showed that the abundance of southern species had increased above levels of the 1950s (Fig. 2). Interestingly, coldwater species persisted despite more frequent failures in recruitment (e.g. Jenkins et al. 2000, Svensson et al. 2005). Resurveys made between 2001 and 2007, largely as part of the Marclim project (www.mba.ac.uk/marclim) and subsequent follow up work, has shown range extensions of several southern species (Mieszkowska et al. 2005). These have occurred at their northern limits in Scotland (e.g. *Gibbula umbilicalis*, Mieszkowska et al. 2006) as well as eastwards into the colder eastern English Channel basin (e.g. *Perforatus perforatus*, Herbert et al. 2003; *Gibbula umbilicalis*, *Patella ulyssiponensis* and *Melaraphe neritoides*, Mieszkowska et al. 2005). These species have breached suspected hydrographic barriers such as Portland Bill, Dorset and St. Catherine's Point, on the Isle of Wight (Crisp & Southward 1958) where

recruitment is low (Herbert et al. 2007), and have jumped across patches of unsuitable habitat, perhaps utilising recently constructed sea defences and marinas as stepping stones (Moschella et al. 2005). Further south in Europe, the southern species of limpet *Patella rustica* has extended its range in northern Portugal past a previous barrier to dispersal as a result of a relaxation of cold water upwelling (Lima et al. 2006), the processes driving this range expansion having been subsequently modelled (Lima et al. 2007a). Other changes have been detected in this region with southern species becoming more persistent (Lima et al. 2006, Lima et al. 2007b).

Other species have made much smaller advances. *Osilinus lineatus* has recolonised areas from which it had disappeared following the cold winter in 1962/63 and has made small advances from these historic range limits in Wales and on the English Channel coast (Mieszkowska et al. 2007). *O. lineatus* has, however, increased in abundance throughout much of its range (Mieszkowska et al. 2007; Fig. 3). *Patella depressa* has made only a small range extension from the Isle of Wight to adjacent sea defences on neighbouring Hayling Island, Hampshire, but again has increased in abundance at many sites on the south and southwest coasts of Britain (Mieszkowska et al. 2005, S. J. Hawkins et al. unpubl. data; Fig. 4). It has, however, not recovered to the levels of the 1950s in North Wales. Limited dispersal capability and suspected specific settlement and nursery ground requirements may be involved in limiting the advances made by these species. Thus, responses are species specific and dependent on life history and other ecological traits.

Although northern species have become less common (e.g. *Semibalanus balanoides* and *Patella vulgata*), major range retractions have not been recorded in the British Isles, except for *Alaria esculenta* in Ireland (Simkanin et al. 2005) and possibly *Tectura testudinalis*, which has not been found on the south coast of the Isle of Man for some years (S. J. Hawkins unpubl. data). Many of these northern species have ranges that extend further south into southern Europe where changes are occurring. Changes in assemblage composition and distribution of macroalgae have been observed with advances of southern species and retraction of northern species (Lima et al. 2007a); *S. balanoides* has also disappeared from the coast of northern Spain (Wetthey & Woodin 2008).

Many of the southern species of animals are able to breed more successfully in warmer years at their northern range limits (Bode et al. 1986, Bowman & Lewis 1986, Lewis 1986, Burrows et al. 1992, O'Riordan et al. 2001, P. J. Moore unpubl.). Warmer, more favourable summers are becoming more frequent and are probably particularly important in the establish-

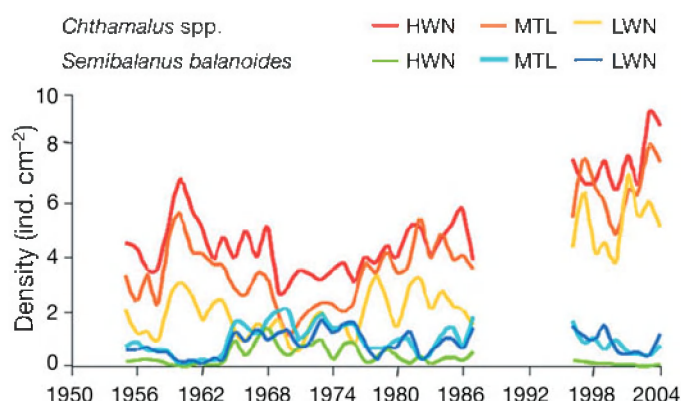


Fig. 2. Long-term changes in northern (*Semibalanus balanoides*) and southern (*Chthamalus* spp.) barnacles averaged for several shores on the south coast of Devon and Cornwall, UK. From Southward et al. (2005). HWN: high water neaps; MTL: mid tide line; LWN: low water neaps

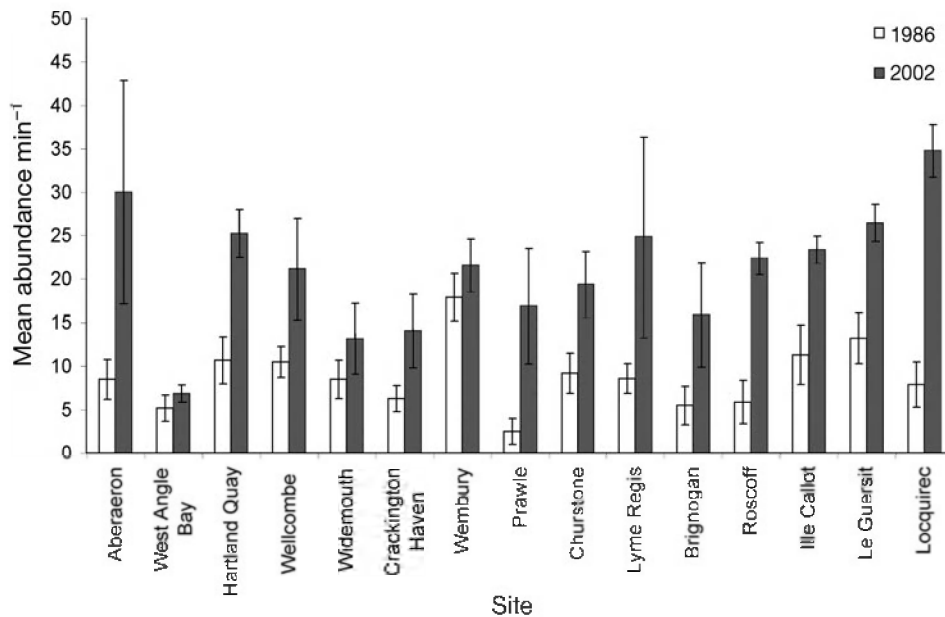


Fig. 3. *Osilinus lineatus*. Mean abundance per minute of timed search at locations in the northern region of its biogeographic distribution immediately prior to the onset of present warming (1986) and 2 decades later (2002). From Mieszkowska et al. (2007)

ment of viable breeding populations and facilitation of range extensions at the northern range limit of southern species. Northern species, however, appear to persist, perhaps because they are better competitors than related southern species (e.g. *Semibalanus balanoides*; Connell 1961b, Poloczanska et al. in press) and occasional massive recruitment events can boost their numbers, especially in species that spawn coincidentally with the spring phytoplankton bloom when good matching can occur (Connell 1961b, Hawkins & Hartnoll 1982, Hansson et al. 2003). Warm springs, however, cause juvenile mortality of *S. balanoides*, releasing *Chthamalus* spp. from competition.

3. COMMUNITY STRUCTURE: LATITUDINAL GRADIENTS IN PATTERN AND PROCESS

In the British Isles, Ireland and northern France, the mid-zone (eulittoral) of sheltered shores is dominated by fucoids which give way to barnacles/mussels and grazing limpets on more exposed shores (Lewis 1964). In northern Europe, fucoids extend further out onto exposed shores; whereas in southern Europe (Spain and Portugal), fucoids are restricted to extreme shelter and estuarine refuges (Ballantine 1961). Classic limpet removal experiments (Jones 1948, Lodge 1948, Southward 1964), follow-up work by Hawkins (1981a,b) and the outcome of major oil spills where all limpets were killed, have all shown that this pattern is directly determined by grazing (Southward & Southward 1978, Hawkins et al. 1992). Subsequent work has confirmed that grazing prevents the establishment of fucoids and other algae throughout much of Europe (Jenkins et al. 2005, Coleman et al. 2006), but wave action determines subsequent survival and persistence (Jonsson et al. 2006).

On moderately exposed shores of the Isle of Man and elsewhere in northern Britain, there are complex direct and indirect interactions between limpets (*Patella vulgata*), barnacles (*Semibalanus balanoides*), fucoids (*Fucus vesiculosus*) (see Hawkins et al. 1992 for review) and microbial films (Thompson et al. 2004), which are delicately balanced leading to patchy and fluctuating distributions of fucoids (Hawkins & Hartnoll 1983b, Hartnoll & Hawkins 1985). The aggregation of *P. vulgata* under adult fucoid patches leads to an

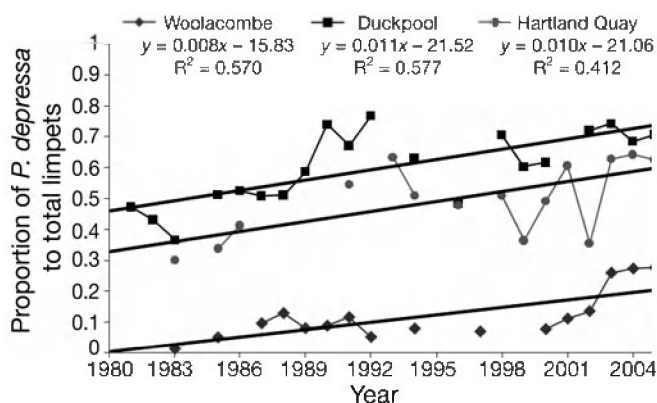


Fig. 4. *Patella depressa*. Proportion of *P. depressa* to total limpets at sites on the north coast of Devon and Cornwall from 1980–2004. Regression analyses are for each site, at $p < 0.01$

uneven grazing field, which can prompt subsequent escapes of juvenile fucoids in areas of reduced *P. vulgata* grazing (Burrows & Hawkins 1998, Thompson et al. 2004; Fig. 5a). Escapes from limpet grazing are crucial to the establishment of patches and are more likely among dense barnacles, in particular *S. balanoides* (Hawkins 1981a, 1983, Hawkins & Hartnoll 1983b). Grazing has also been shown to prevent furoid and additional macroalgal escapes on barnacle-covered shores throughout Europe (Coleman et al. 2006). In southwest Britain and southern Europe, however, a

reduction in grazing pressure is less likely to lead to algal escapes and dense patches of fucoids as occurs on more northern shores (Jenkins et al. 2005). These patterns have been examined spatial statistics (Johnson et al. 1998b) and process simulated using models (Burrows & Hawkins 1998, Johnson et al. 1998a). These techniques have also highlighted the importance of aggregative behaviour of *P. vulgata* under furoid patches in the dynamics of the system as well as the role of stochastic events such as barnacle, furoid and limpet recruitment.

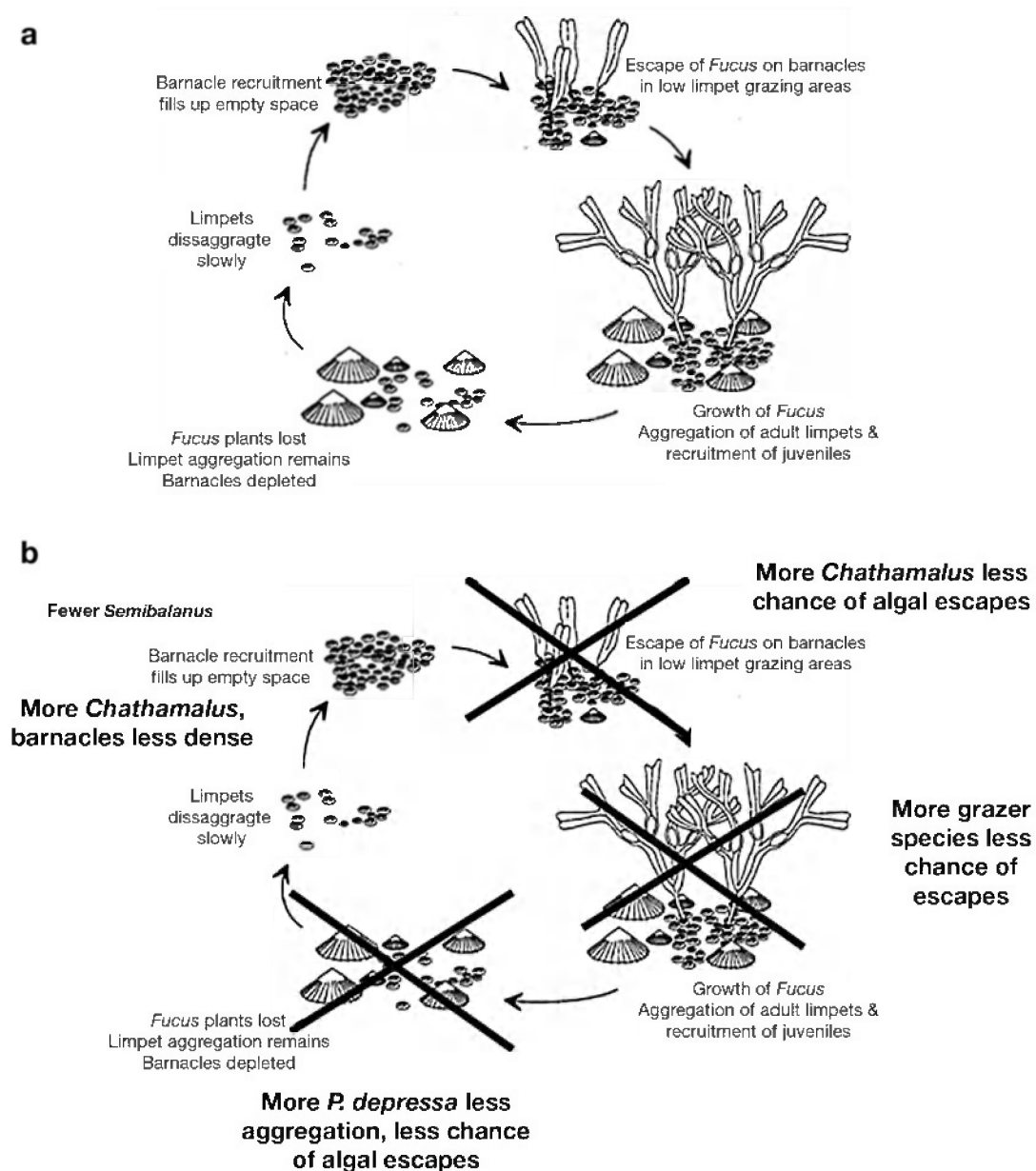


Fig. 5. Sequence of events on patchy, moderately exposed shore over several years. (a) Shores dominated by *Patella vulgata* and *Semibalanus balanoides*, (b) shores dominated by *P. depressa* and *Chthamalus* spp. Adapted from Burrows et al. (1998)

With climate change, the diversity of grazers will increase in northern Europe. Similarly, large barnacles such as *Semibalanus balanoides* capable of extremely dense settlement will be replaced by smaller and less rapidly growing chthamalids (Connell 1961b, Southward 1991, Poloczanska et al. in press). The rates and consequences of such changes are explored in the next section.

4. FORECAST AND PREDICTION: FUTURE STRUCTURE AND FUNCTIONING

Bioclimatic envelope approaches have been used to forecast future distributions of various organisms (see Pearson & Dawson 2003 for a review of different climate envelope methods). These approaches offer a good first-order approximation, giving the likely outer limits of potential distribution through changes in the geographic extent of the potential niche of a species (Pearson & Dawson 2003). These approaches have, however, been criticised because they neglect important factors such as dispersal, habitat quality and connectivity, and biological interactions (Davis et al. 1998, Ibanez et al. 2006). In intertidal systems, experimental approaches have shown that simple models are often unable to predict patterns in species distributions and abundance due to the complex nature of physical and biological interactions across latitudinal gradients (e.g. Helmuth et al. 2002, Sagarin & Somero 2006). We have used a climate envelope approach for intertidal climate indicator species and have found that by using wave exposure (see Burrows et al. 2008) and February isotherms, a reasonable predictive capability is possible (M. T. Burrows et al. unpubl.). The use of long-term data sets can increase the robustness of modelling approaches by testing modelled simulations using hind-cast techniques. These techniques allow modelled data to be compared against empirically collected data to test the model fit. Forecasts of future states are also possible using published climate change scenarios and our modelling shows the potential for range extensions of many species. For the English Channel, we predicted range extensions before they were found in the field for *Gibbula umbilicalis*, *Patella ulysiponensis* and *Melaraphe neritoides* (M. T. Burrows et al. unpubl. data, S. J. Hawkins et al. unpubl. data).

A more analytical approach is to build biological interactions into model predictions. The barnacle data set collected by Southward since the 1950s, experiments on the role of competition (e.g. Connell 1961b), and good models of the dynamics of open populations (Roughgarden et al. 1988, Roughgarden et al. 1994, Connolly & Roughgarden 1999, Connolly et al. 2001) have enabled a mechanistic approach to the prediction

of future populations (Poloczanska et al. in press). Detailed path analysis has shown that while temperature in the early summer explains the abundance of the northern barnacle *Semibalanus balanoides* extremely well, it does not explain the abundance of the southern barnacle species *Chthamalus*. This is best explained as an indirect effect mediated by release from competition with *S. balanoides* in warmer years. A variety of models have been constructed, but the best fit to the 40 yr time series involved a space-limited model simulating interference competition between *S. balanoides* and juvenile *Chthamalus* spp. Having constructed a workable model, predictions of future trends have been made using published UKCIP02 (United Kingdom Climate Impact Programme) climate change scenarios. The predictions using both high and low future emissions scenarios suggest that *S. balanoides*, the dominant barnacle in the southwest of England during the 1930s (Moore 1936), will become locally extinct in the next 25 to 50 yr (Poloczanska et al. in press).

Experiments have also been undertaken to explore the consequences of changes in species identity of the major grazers *Patella* spp. (Moore et al. 2007a, Moore et al. 2007b). This work confirms that the northern species of limpet *P. vulgata* preferentially aggregates under fucoid patches (Moore et al. 2007a, P. J. Moore unpubl. data, see also Hawkins & Hartnoll 1983a,b). If these patches are removed, significant numbers of *P. vulgata* relocate their home scars beneath new fucoid patches or die (Moore et al. 2007a; Fig. 6). In contrast, the southern limpet species *P. depressa* does not preferentially aggregate beneath *Fucus* patches; if found beneath *Fucus*, it does not respond when the canopy is removed (Moore et al. 2007a). Should *P. depressa* continue to increase in abundance, then the cycle of aggregative behaviour typical on shores such as the Isle of Man (Hartnoll & Hawkins 1985) where *P. vulgata* is the only species, will not occur. Thus, the shore is likely to be less patchy (Johnson et al. 1998a) and also less dynamic (Burrows & Hawkins 1998). Furthermore, smaller and slower growing barnacles (*Chthamalus* spp.) are less likely to promote fucoid escapes typical on *Semibalanus balanoides* dominated shores (Fig. 5b).

Although so far untested, the addition of extra grazers such as trochids (*Gibbula umbilicalis* and *Osilinus lineatus*) and decapods (*Pachygrapsus marmoratus*) will make escapes of *Fucus* and other algae less likely. Greater physical stress will also slow early growth of fucoids reducing the probability of escapes (Thompson et al. 2004). Thus, shores are likely to have less primary producing fucoids which will be increasingly restricted to sheltered shores. Mid-latitude shores such as those in southern Britain and Brittany, France may resemble those in Portugal in the foreseeable future. Primary production will decline and there will be less export of

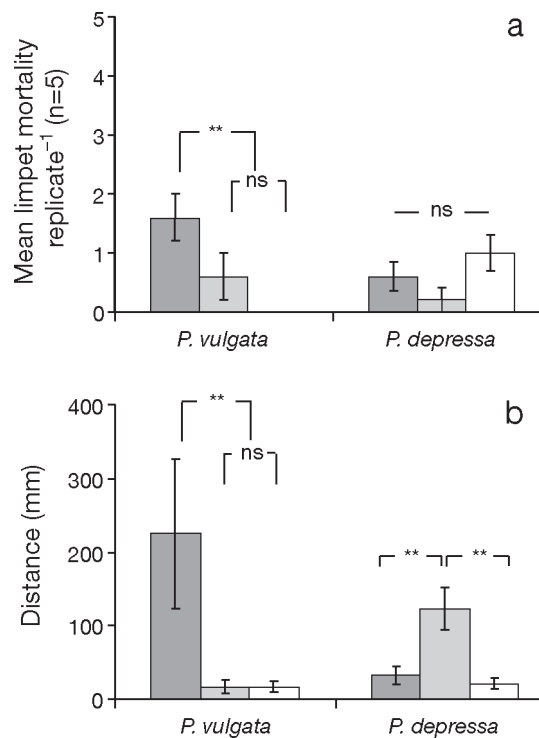


Fig. 6. *Patella vulgata* and *P. depressa*. Response to experimental loss of *Fucus vesiculosus*. (a) Mortality, (b) distance moved to a new home scar. Means \pm 1 SE; ** $p < 0.01$, ns: not significant. From Moore et al. (2007a)

macroalgal detritus into coastal ecosystems. Secondary production will also be reduced as *Chthamalus* spp. grows much more slowly than *Semibalanus balanoides*. Thus, climate change is likely to have considerable consequences on the functioning of coastal ecosystems.

There are, however, some surprises in store due to both lags in responses of different species and idiosyncratic trait-dependent differences in the role of species. For example, the southern species of hermit crab *Clibanarius erythropus* that previously colonised Britain during the warm period experienced in the 1950s has yet to return (Southward & Southward 1977, 1988, S. J. Hawkins pers. obs.). The recolonisation and recovery of *C. erythropus* may have been further slowed by interaction with pollution (Southward & Southward 1978). The English Channel has been subjected to repeated pollution incidents (e.g. the tankers 'Torrey Canyon', 'Amoco Cadiz') which not only decimated British populations, but also reduced potential sources of recruits in northern France. Tributyl tin (TBT) pollution also reduced dogwhelk populations with recovery only occurring in the late 1990s, following the ban on TBT-based paints on smaller vessels. Thus, the favoured shell of *C. erythropus* also became rarer. In addition, northern species which retreated in the 1950s such as *Alaria esculenta*, never really recov-

ered during the subsequent cold period of the 1960s, 1970s and early 1980s. Moreover, recent work has shown that the southern species of limpet, *Patella depressa*, does not control macroalgal abundance to the same degree as the northern species *P. vulgata* (Moore et al. 2007b). Fucoid escapes were able to occur in enclosures containing just *P. depressa* as the grazing activity of this species was significantly reduced during early spring to late summer when their gonads were full. During this time, *Fucus* germlings settled on the shore and were able to grow to a size less vulnerable to the grazing activities of *P. depressa*, which increased in autumn (Moore et al. 2007b). In contrast, in enclosures with only *P. vulgata*, no *Fucus* escapes were found and there was no reduction of grazing activity during the settlement period of *Fucus*. These results were surprising and indicate that as species identities change in response to climate change, there could be considerable impacts on the way species interact due to subtle differences in their behaviour or physiology. Changes in the strength or direction of species interactions could fundamentally alter assemblage composition with consequences for ecosystem functioning. The changes we have described are likely to have consequences for the balance and dynamics of fucoids on European rocky shores and hence on whether a particular shore is a net importer or exporter of material. Such changes in fucoid cover will also impact biodiversity given the number of mobile and sessile species that live in and among fucoids (Thompson et al. 1996).

5. FUTURE WORK: PATTERN, PROCESS AND PREDICTION

Although there is a rich heritage of monitoring responses of species to climate fluctuations in the NE Atlantic, there are still many areas that need further work. The underlying mechanisms driving change that link individual ecophysiology with reproductive success and hence population dynamics are not well understood (but see Sagarin & Somero 2006). Knowledge of the effect of species interactions on species responses to climate change is still limited (but see Helmuth et al. 2006, Moore et al. 2007a, Moore et al. 2007b, Blight & Thompson 2008). Work to date indicates that species interactions will alter under future climate change scenarios, with implications for community structure (Moore et al. 2007a, Moore et al. 2007b, Sanford & Swezey 2008) and ecosystem functioning. In addition, most field and laboratory experiments investigating responses of species to climate change have manipulated mean values in an environmental variable (e.g. temperature) with few studies

manipulating the amount of variability in environmental conditions (Benedetti-Cecchi 2003). This is surprising considering that extreme events are predicted to become more frequent with climate change (IPCC 2007) and that the incorporation of spatial and temporal variance is vital for understanding ecological processes (Benedetti-Cecchi 2000, 2003, Benedetti-Cecchi et al. 2005).

As with many components of marine biodiversity, rocky shore species are showing changing abundance and geographic range in response to climatic warming. At present, more advances than retreats have been recorded. In northern species, this is perhaps due to the greater reproductive output of single brood spring spawners and greater competitive ability. This hypothesis, however, remains to be tested. It must also be remembered that more studies have been made in regions where species reach their northern limits than in lower latitudes where many southern limits are located, potentially skewing evidence. Assemblage-level monitoring needs to be undertaken to measure the effects of changes in the balance of canopy algae and suspension feeders over the next 50 yr and hence the potential contribution of rocky shores to inshore productivity. In addition, experimental studies on population- and community-level processes need to be undertaken to enable modelling techniques that provide better predictions. Better understanding will enable progression from mere forecasts to better predictive modelling and future studies under different climate scenarios. Differences between modelled and actual distributions can then be used to generate testable hypotheses about the processes setting geographic limits, particularly the interaction of ultimate factors such as temperature regime with proximate factors such as habitat quality, connectivity of habitat patches, dispersal capability, the porosity of hydrographic barriers such as major headlands and the role of positive and negative biological interactions.

Rocky shore species have long been known to provide excellent sentinels for detecting changes in biodiversity in other systems that are less accessible and more expensive to survey, such as offshore systems (Lewis 1986, Southward et al. 1995, Southward et al. 2005). Rocky shores also provide a tractable system for field experiments to explore the likely community structuring and ecosystem functioning consequences of changes in species distributions and assemblage composition. Existing time series need to be continued and resurveys made on a European scale to fully exploit the classical baselines provided by Fischer-Piette, Crisp and Southward (e.g. Fischer-Piette 1936, 1955, Fisher 1943, Crisp 1950, 1964, Southward 1951, 1958, 1967, 1980, 1991, Southward & Crisp 1952, 1954a,b, 1956, Crisp & Knight-Jones 1954, Fischer-Piette & Prenant

1956, 1957, Crisp & Southward 1958, Crisp & Fischer-Piette 1959, Fischer-Piette & Gaillard 1959, Southward et al. 1995, Southward et al. 2005).

6. INFORMING ADAPTATION

There are major policy implications of climate change in marine ecosystems from a societal perspective. The inertia of the climate system means that society must live with continuing climate change over the next 50 yr or so (IPCC 2007), until mitigation measures and new technologies enable a switch from a carbon-based economy. Adaptive policies need to be based on real-time monitoring of changes to distinguish actual trends from mere fluctuations. Knowledge of future states would also enable a more precautionary approach to managing the interactions of global change and local- and regional-scale impacts.

Comprehensive, broadscale and long-term observations are essential for a sufficiently accurate knowledge of the state of our seas and coasts to allow adaptive management. Sustained observation/monitoring alone is not enough: integrative experimental studies are required to understand the mechanisms involved in individual and population responses to climate change.

Adapting to climate change requires disentangling human-driven global change of low amplitude and frequency from natural temporal and spatial fluctuations as well as human impacts acting at regional (e.g. overfishing, eutrophication, non-native species) and local scales (aquaculture, habitat loss due to inappropriate coastal development, point-source pollution). Some local impacts can scale up to whole coastlines, such as that of low-crested sea defences (Airoldi et al. 2005) resulting in regional scale habitat alteration and loss. Adaptive policies must focus on managing the interactions of global change (which we cannot slow in the short to medium term, i.e. next 50 yr), with drivers that society can influence and ameliorate, such as overexploitation, pollution and habitat degradation and loss.

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