

Review of climate change impacts on marine aquaculture in the UK and Ireland

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ABSTRACT

1. Marine aquaculture relies on coastal habitats that will be affected by climate change. This review assesses current knowledge of the threats and opportunities of climate change for aquaculture in the UK and Ireland, focusing on the most commonly farmed species, blue mussels (*Mytilus edulis*) and Atlantic salmon (*Salmo salar*).

2. There is sparse evidence to indicate that climate change is affecting aquaculture in the UK and Ireland. Impacts to date have been difficult to discern from natural environmental variability, and the pace of technological development in aquaculture overshadows effects of climatic change. However, this review of broader aquaculture literature and the likely effects of climate change suggests that over the next century, climate change has the potential to directly impact the industry.

3. Impacts are related to the industry's dependence on the marine environment for suitable biophysical conditions. For instance, changes in the frequency and strength of storms pose a risk to infrastructure, such as salmon cages. Sea-level rise will shift shoreline morphology, reducing the areal extent of some habitats that are suitable for the industry. Changes in rainfall patterns will increase the turbidity and nutrient loading of rivers, potentially triggering harmful algal blooms and negatively affecting bivalve farming. In addition, ocean acidification may disrupt the early developmental stages of shellfish.

4. Some of the most damaging but least predictable effects of climate change relate to the emergence, translocation and virulence of diseases, parasites and pathogens, although parasites and diseases in finfish aquaculture may be controlled through intervention. The spread of nuisance and non-native species is also potentially damaging.

5. Rising temperatures may create the opportunity to rear warmer water species in the UK and Ireland. Market forces, rather than technical feasibility, are likely to determine whether existing farmed species are displaced by new ones.

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INTRODUCTION

Aquaculture is the cultivation of aquatic organisms, usually for the purposes of human consumption. It implies a level of intervention in the rearing process to enhance survival rates and production through a combination of regular feeding, the provision of substratum or through protection from predators (FAO, 2002). Freshwater, brackish and seawater environments are all exploited for aquaculture, ranging from hatcheries located on freshwater rivers, to estuaries, fjords, and open bays.

There is a close relationship between prevailing environmental conditions and the success of aquaculture production. This is largely because the health of both finfish and shellfish is heavily dependent on environmental conditions, such as temperature, salinity, oxygen solubility and dissolved waste products (Mydlarz *et al.*, 2006). Additionally, the physical processes of waves, tides, rivers, and associated erosion or deposition may alter the suitability of the abiotic environment. The distribution of a specific species is limited by its tolerance range to local environmental variables. Towards the edge of its tolerance ranges, a species may become immuno-compromised, such that it becomes more susceptible to disease or to predation. In addition, changes in the environment may have direct or indirect effects on the abundance of parasites and pathogens (Karvonen *et al.*, 2010), or the success of predatory and competitor species, either native or introduced.

The success of aquaculture is fundamentally dependent on the complex ecology of the aquatic environment, which is a major concern considering the wide ranging impacts of climate change (IPCC, 2007). Aquaculture facilities are often located in areas which are likely to bear the brunt of climate change impacts: coasts and estuaries are susceptible to changes in water temperature, storm intensity or frequency and sea level. On a more complex level, shifts in the aquatic ecosystem will impact upon relationships between prey, predators, parasites, and pathogens. In rivers and estuaries, changes to flood frequency and magnitude are an additional concern, especially because of the sediments, nutrients, pathogens and contaminants associated with run-off.

Extent, value and types of marine aquaculture in the UK and Ireland

Marine aquaculture in the UK and Ireland can be broadly divided into the shellfish and the finfish

sectors (Figure 1, Tables 1 and 2). Finfish aquaculture sites are clustered towards the north of the UK on the Western Scottish Highlands, Argyll, the Hebrides and Northern Isles. The western coasts of Ireland and Northern Ireland are characterized by a high concentration of shellfish sites as well as significant finfish production, while in England and Wales shellfish aquaculture predominates (Figure 2).

Scotland leads marine finfish aquaculture, producing over 90% of the UK and Ireland's marine finfish (Table 1; BIM, 2009; Cefas, 2011a). Ireland is the second largest finfish producer. While freshwater fish are cultivated in Northern Ireland and England, marine finfish production is negligible. In 2009, no marine finfish production was recorded in offshore areas for Wales.

Ireland produces almost half of the UK and Ireland's shellfish (Table 2; BIM, 2009; Cefas, 2011b), followed by Wales with 20%. Northern Ireland, England and Scotland share the remaining 30% fairly evenly.

Dominant species cultivated vary from region to region. In Scotland, Atlantic salmon (*Salmo salar*) accounts for 98% of Scottish marine finfish aquaculture at an estimated annual economic value of £ 412 million (Scottish Government, 2010). In addition to Atlantic salmon, small volumes of halibut (*Hippoglossus hippoglossus*) and rainbow trout (*Oncorhynchus mykiss*) are also produced in offshore areas (Table 1). In terms of shellfish, the Scottish industry is dominated by mussels (*Mytilus edulis*), followed by lesser volumes of Pacific oysters (*Crassostrea gigas*), native oysters (*Ostrea edulis*), queen scallops (*Aequipecten opercularis*) and scallops (*Pecten maximus*).

In contrast, Irish aquaculture produces far more shellfish than marine finfish. Mussels accounted for almost 80% of Irish shellfish production, followed by Pacific oysters, native oysters, clams and scallops (Table 2). Finfish produced were primarily Atlantic salmon (96.23%) in addition to small volumes of sea reared trout (3.77%). Although Ireland produces more shellfish than finfish, the finfish sector has a higher economic value. Irish finfish was valued at €72.0 million compared with €34.6 million for shellfish (BIM, 2009).

England, Northern Ireland and Wales produce very little marine finfish, and instead focus on onshore rearing of freshwater finfish species such as rainbow trout (Cefas, 2011a). As freshwater species and onshore aquaculture are not the focus of this review, they have been omitted here. In



Figure 1. In the UK and Ireland, mussels (*Mytilus edulis*) and Atlantic salmon (*Salmo salar*) are the most commonly farmed marine species. Conventional cultivation and harvesting techniques are illustrated; (a) mussel dredge; (b) mussel raft; (c) salmon pens; (d) seaweed aquaculture (*Saccharina latissima*) provides potential for expansion of the UK and Ireland aquaculture industries. (Source of images: Ruth Callaway a, b, c; Michele Stanley d.)

Table 1. Marine finfish production in the UK and Ireland (BIM, 2009; Cefas, 2011a)

	Total volume (tonnes) by region					Total production by species (tonnes)	Percentage of total production by species (%)
	England	Wales	Scotland	Northern Ireland	Ireland		
Turbot	1	1	< 0.01
Halibut	189	189	0.1
Atlantic salmon	144 247	407	12 210	156 864	97.9
Rainbow trout or sea reared trout*	2620	...	478	3098	1.9
Total production by region (tonnes)	1	0	147 056	407	12 688	160 152	
Percentage of total production by region (%)	< 0.01	0.0	91.8	0.3	7.9		

*This figure includes rainbow trout produced in cages in sea water in offshore areas, as well as Irish sea reared trout.

Table 2. Marine shellfish production in the UK and Ireland in 2009 (BIM 2009; Cefas, 2011b)

	Total volume (tonnes) by region					Total production by species (tonnes)	Percentage of total production by species (%)
	England	Wales	Scotland	Northern Ireland	Ireland		
Pacific oyster	811	4	232	309	6,488	7,844	11.3
Native oyster	54	0	39	127	358	578	0.8
Scallops	0	...	4	...	55	59	0.1
Queen scallops	6	6	0.0
Mussels	3800	13 812	6302	8015	26 502	58 431	84.5
Clams	13	1	162	176	0.3
Cockles	2027	2027	2.9
Total production by region (tonnes)	6705	13 816	6583	8452	33 566	69 122	
Percentage of total production by region (%)	9.7	20.0	9.5	12.2	48.6		

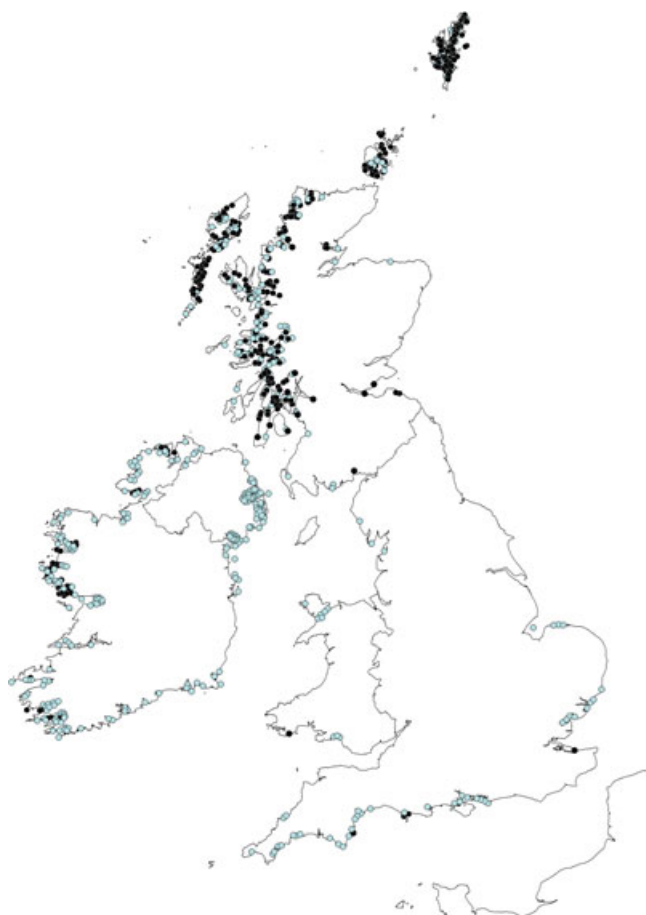


Figure 2. Geographical distribution of marine shellfish (blue symbols) and finfish (black symbols) aquaculture production facilities in the UK and Ireland.

these parts of the UK marine aquaculture is shellfish dominated, with mussels representing the greatest proportion of production in all three regions (Table 2).

Aim of review

The aim of this review is to establish the state of knowledge with regards to the impact of climate change on marine aquaculture in the UK and Ireland, and to highlight areas where further research is required to inform future policy and decision-making (Figure 3). This review, initiated by the UK Marine Climate Change Impact Partnership (MCCIP), considers UK and Irish aquaculture industries that utilize coastal habitats to farm marine species for human consumption, with a particular focus on Atlantic salmon and blue mussel farming.

The level of human intervention varies between farming processes. For instance, mussel farming can be a relatively low intervention technique limited to the relocation or settlement of wild spat

at sites suitable in terms of food supply and shelter from storm and wave damage. There is though an increasing requirement for management in rope cultured mussel production as sites increase in size and stock management becomes more sophisticated to optimize growth, minimize fouling and maximize spat fall. Salmon farming in pens or cages is tightly managed as stock densities and feeding is controlled, and fish are given protection from predators. Although farmed salmon are not dependent on natural resources, successful aquaculture is completely reliant on the abiotic environment in which it is conducted. Marine species that are reared in land-based recirculating aquaculture systems are not directly affected by climate change and have not been considered in this review.

The review begins by assessing the impact climate change will have on culture environments, and investigates what is known about effects of ocean acidification. The physiology, health and epidemiology of aquaculture species is then discussed against a backdrop of environmental change. The review also highlights how interactions between nuisance and harmful species and aquaculture species could change in future climates.

While this review focuses on the direct effects of climate change on aquaculture, it is acknowledged that indirect effects play an important role. The aquaculture industry is driven by a complex set of economic, operational and socio-economic parameters, ranging from market demand to financial support, cost of energy, state of wild fish stocks and land and property values (Young, 2001; Whitmarsh and Palmieri, 2008). Many of these factors are influenced by climate change and will in turn affect the progress of the UK and Irish aquaculture industry. It is beyond the scope of this paper to deal with all of these issues, but given the importance of fish feed supply for aquaculture, the last section of this review is dedicated to the subject of global fishmeal and fish oil resources.

IMPACT OF CLIMATE CHANGE ON ENVIRONMENTS SUITABLE FOR AQUACULTURE

Marine aquaculture sites generally make use of naturally existing coastal habitats that are suitable for the target species. As such, species accustomed to sea water and the daily rhythm of tides and



Figure 3. Climate change can potentially affect many aspects of the UK and Ireland aquaculture industries; panels (a)–(e) illustrate some of the impacts: (a) jellyfish blooms (*Pelagia noctiluca*) can kill farmed fish; (b) the non-native Pacific oyster (*Crassostrea gigas*) colonizes wild mussel beds; (c) blooms of harmful microalgae can cause shellfish poisoning; (d) the development of mussel larvae may be affected by ocean acidification; (e) climate change affects wild fish populations that are used in fish feed (source of images: Feargus Callagay a; Kerstin Kolbe b; Ruth Callaway c, e; Robert Ellis d).

waves are typically grown in estuaries and shallow coastal waters. Mussels may be farmed from relocated spat or grown on racks or suspended ropes, while salmon are reared in cages or pens. The manner in which salmon and mussels are cultivated exposes them to risks associated with changes in ocean temperature, wave regime, storm frequency or sea level.

Sea-level rise

Relative sea-level rise (RSLR) is the actual rate of sea-level rise adjusted to include the impact of isostatic change. The current relative rate of sea-level rise in the south-west of England is about 2.5 mm yr^{-1} , while in Scotland sea-level rise attributed to climate change is almost offset by uplift, with RSLR values of between 0.1 and 0.9 mm yr^{-1} (Shennan and Horton, 2002; Shennan *et al.*, 2006). The rate of sea-level change in Wales and the south-east of England is between these values.

Over the next century, the effect of sea-level rise on habitats suitable for aquaculture will be spatially variable. In areas of isostatic uplift the effect is minimized, while towards the south of the UK subsidence enhances the impact of rising sea levels. However, rates of sea-level rise are likely to accelerate. Jenkins *et al.* (2009) indicate that the

projected range of absolute sea-level rise around the UK, excluding land movements, is between 12 and 76 cm by 2095. This suggests an 8-fold increase in current rates of sea-level rise toward the turn of the century. At a rate approaching 1 cm yr^{-1} sea-level rise will easily outstrip rates of isostatic uplift throughout the UK and Ireland.

One of the greatest challenges facing aquaculture in coastal settings is understanding how coastal habitats will respond to accelerating sea-level rise, particularly as the shoreline morphology shifts. It is currently assumed that the rate of coastal habitat retreat will increase as the RSLR increases, but this view may be challenged, with some research indicating that the reorganization of the shoreline has been sluggish during periods of rapid sea-level rise in the past (Orford and Pethick, 2006). As a result, habitat types such as intertidal mud flats, which are lost due to sea-level rise, may not necessarily be replaced landward.

Of additional concern is the lack of sediment available for reformation of coastal morphologies at landward locations, impeding coastline recovery and increasing coastal habitat vulnerability to wave erosion (Orford and Pethick, 2006). In estuaries, for instance, maintenance of depth as sea level rises requires fine sediment deposition to occur at an equal rate. If sufficient fine sediment is

not available, intertidal surface elevations will fall, reducing the area for shellfish species such as cockles (*Cerastoderma edule*) and mussels. However, deeper inlets may be more suitable for mussel lays.

In addition to the rapid rate of sea-level rise expected towards the end of the century and the possibility of an insufficient supply of sediment to redeposit coastal landforms, the assumption of coastal habitats uniformly retreating is challenged by 'coastal squeeze'. Developments focused on the coast provide a hard edge to retreating habitats, and as a result, there may not be adequate space for habitats to retreat landward. Crooks (2004) suggested that increases in sea level will result in the loss of intertidal habitats if flood defences are maintained, unless a position of managed retreat is adopted. Habitat loss in intertidal areas is likely to impact shellfish cultivations, while finfish aquaculture may be less affected. More research is required regarding coastal sediment budgets and the impact of accelerated sea-level rise on coastal morphology and associated habitats.

Storm damage

Over the last 50 years there has been an intensification of storm events in the north-east Atlantic (Jones *et al.*, 1999; Alexander *et al.*, 2005; Yan *et al.*, 2006). The trend is predicted to continue (Leckebusch *et al.*, 2006), although the rate of increase may be slow (Weisse *et al.*, 2005). Extreme wind speeds and wave conditions are likely to increase concurrently (Grabemann and Weisse, 2008). The severity of storms, their frequency, the height of waves and storm surges are all predicted to increase over the coming decades (Frost *et al.*, 2012).

Storm damage is a major concern to the aquaculture industry. Mussel cultivations are vulnerable to destruction by storms (Dankers, 1995), and in intertidal areas storms are a major factor limiting the distribution of mussel beds to sheltered parts (Nehls and Thiel, 1993).

A study of fish escapes in Scotland found that of the 2.18 million fish that escaped during the seven years covered by the research, 38% escaped during a single storm event in 2005 (Taylor and Kelly, 2010). Similarly, during a summer flood in Boscastle, southern England, one owner lost 40 000 salmon and brown trout (*Salmo trutta*) (Handisyde *et al.*, 2006). Storms cause large waves, surges and flooding, all of which can potentially result in structural damage to

infrastructure or the introduction of predators (Handisyde *et al.*, 2006). This leads to a loss of stock, higher capital cost required to design infrastructure that can withstand large events and increased insurance costs.

Failure of cages and moorings is most likely to occur during unsettled wave and wind conditions, prompting Taylor and Kelly (2010) to suggest that cages were under-designed for the environmental conditions to which they were exposed. Storm damage to marine salmon cages poses financial and welfare problems, as well as providing a route for farmed salmon to escape into the environment and interact with wild fish, causing hybridization and loss of genetic diversity (Walker *et al.*, 2006). A greater frequency of extreme weather events is also anticipated to facilitate disease epidemics by causing ecological stress and thereby compromising resilience (Epstein, 2001). However, on the positive side, extreme weather events can directly harm and reduce parasite populations through environmental disturbance (Overstreet, 2007). Information regarding the impact of climate change on storm surges, wave height and wave frequency should be heeded when designing cages for a specific return storm period.

The systematic recording of incidences of fish escapes began in 2002 for Scottish fish farming, and no similar records exist for regions outside of Scotland. As such, it cannot be determined as to whether the number of fish lost has increased, whether incidents of escape are significantly correlated with storm incidence, or whether the number of fish that escape correlates with storm intensity. The greatest economic losses could potentially occur in Scotland owing to the concentration of salmon farming in this region. However, the Scottish Government is in the process of developing legislated engineering standards for cages, and combined with a risk analysis of previous incidents, this impact may be mitigated in the future.

Precipitation, freshwater input and associated pollution

Evidence suggests that the intensity of precipitation events will increase in the winter months, while summers may become drier (Frost *et al.*, 2012). Lowe *et al.* (2009) predict that in the northern UK where the majority of finfish aquaculture takes place, there will be a reduction in summer rainfall of up to 30% by 2098 compared with 1961–1990 values. An altered hydrology, characterized by

more frequent floods and droughts, will have a complex effect on the transport and concentration of toxicants, nutrients and pollutants. This is illustrated by a study in Portugal, which found that during periods of lower than average rainfall, episodes of diarrhetic shellfish poisoning increased owing to reduced estuarine mixing (Vale and de Sampayo, 2003).

Conversely, increases in peak river discharge during winter are likely to present a major problem to infrastructure developed in rivers and estuaries. An increase in rainfall intensity during winter will result in enhanced run-off and higher flood discharge peaks (Roessig *et al.*, 2004; Harley *et al.*, 2006). The impact of increased run-off on water quality is three-fold: increased turbidity, increased nutrient loads and increased contaminant loads. Extreme precipitation events and flooding are often associated with significant nutrient pulses to the near shore environment (Devlin and Brodie, 2004), frequently as a result of sewage overflows (Kay, 2008). Nitrogen-related pollution of coastal waters has caused widespread hypoxia and anoxia, habitat degradation, alteration of food-web structure and loss of biodiversity (for review, see Howarth, 2008). Data collected by the Scottish Environment Protection Agency suggest that organic carbon levels in Scotland have doubled within the last 20 years as a response to increases in run-off (Sheahan *et al.*, 2010).

The problem is likely to be severe for aquaculture facilities located in the estuaries of rivers with large catchments. Following an outbreak in the human population, the norovirus may pass through the sewage system. Heavy rainfall (or snow meltwater) often results in tertiary treatment works being bypassed, resulting in the discharge of effluent in a raw or highly contaminated state. The Shellfish Association of Great Britain has seen an increase in the number of incidents of norovirus associated with bivalves; in 2009 there were 12 outbreaks with 24 positives from bivalve shellfish affecting 739 people (Pickerell, 2010).

The Intergovernmental Panel on Climate Change (IPCC) (2007) suggest that the global oceans are freshening. However, there is substantial uncertainty and a lack of data concerning macro-scale and long-term changes in salinity within UK coastal waters as a result of climate change and the potential impacts on UK aquaculture are even less defined. For UK aquaculture, the most pressing needs are for a long-term monitoring programme that studies

salinity changes at aquaculture sites, as well as an enhanced understanding of the impacts of salinity changes on UK aquaculture species.

OCEAN ACIDIFICATION

Ocean acidification is a recognized threat to marine ecosystems (Frost *et al.*, 2012). With atmospheric carbon dioxide (CO_2) predicted to rise to between 550 ppmv and 958 ppmv by 2100, ocean pH is predicted to fall by 0.3–0.5 units and carbonate saturation states are projected to decline by about 45% before the end of this century (IPCC, 2007; Andersson *et al.*, 2008). Coastal and estuarine ecosystems are expected to experience effects of acidification earlier and more severely than other systems (Orr *et al.*, 2005; Feely *et al.*, 2008, 2010; Thomsen *et al.*, 2010; Range *et al.*, 2011). As marine aquaculture industries utilize these ecosystems for production, there are potentially significant socio-economic implications.

At present, few studies have measured the biological effects as a result of the current decline in ocean pH and carbonate ion concentration from the pre-industrial levels (Talmage and Gobler, 2010). There has, however, been a significant amount of research over the last decade to determine the biological effects that ocean acidification will cause by 2100. The response to these levels of acidification varies between taxa, with recent studies demonstrating inter-specific variation between closely related species (Miller *et al.*, 2009), as well as intra-specific variation both between and within populations (Parker *et al.*, 2011). Particular vulnerability has been observed in the majority of marine calcifying groups, principally through the reduced ability to produce and maintain calcium carbonate (CaCO_3) structures under the declining ocean carbonate ion concentrations. As such, the predominant impact of ocean acidification on the marine aquaculture industry is expected to occur in these marine calcifying groups, primarily affecting the marine shellfish aquaculture industries (Cooley and Doney, 2009).

The principal biological effect is likely to occur during the vulnerable early developmental stages of bivalves; this is due to marine bivalves producing a soluble amorphous CaCO_3 shell as a transient precursor to the aragonite and calcite shells that are secreted by adults (Weiss *et al.*, 2002; Fabry *et al.*, 2008). This form of CaCO_3 is

significantly less stable than the crystalline phases of aragonite and calcite, rendering it more difficult for larvae to produce their shells under the continuing carbonate decline (Raven *et al.*, 2005). Further, it has been demonstrated that smaller post-larval bivalves are less able to overcome dissolution pressure at low pH (Waldbusser *et al.*, 2010). Subsequent impacts on shell formation can lead to bivalves developing growth abnormalities (Kurihara, 2008; Parker *et al.*, 2010), declining growth (Michaelidis *et al.*, 2005; Talmage and Gobler, 2009, 2010; Gazeau *et al.*, 2010; Bechmann *et al.*, 2011) and reduced survival rates (Watson *et al.*, 2009; Talmage and Gobler, 2010). In the wild, bivalve larvae naturally exhibit a high mortality rate (>98%) during their transitional phase from free swimming larvae to juveniles, and therefore any additional stresses as a result of ocean acidification could significantly reduce the number of individuals recruited (Green *et al.*, 2004). An example of this has already been demonstrated in the wild, with significantly lower bivalve recruitment occurring at naturally low pH sites (Cigliano *et al.*, 2010). The supply of viable spat could therefore be compromised by ocean acidification conditions, which would potentially impact mussel farms around the UK and Ireland. This could force mussel farms to either rely more heavily on hatchery produced seed in the future, or to utilize a different mussel species that is more tolerant to ocean acidification (Cooley and Doney, 2009; Bell *et al.*, 2010).

Older life stages of bivalves have also shown to be affected by the predicted future climate scenarios. Adverse effects on calcification rates, shell dissolution and the disruption of the internal acid–base balance have been demonstrated. The latter has been associated with a decreased metabolic rate, declined tissue growth and suppressed immune responses (Barry *et al.*, 2005; Michaelidis *et al.*, 2005; Berge *et al.*, 2006; Bibby *et al.*, 2007; Gazeau *et al.*, 2007; Beesley *et al.*, 2008; Wood *et al.*, 2008).

Selective breeding, such as that in the oyster *Saccostrea glomerta* (Parker *et al.*, 2011), could potentially be used to obtain brood stock with an increased tolerance towards the impacts of ocean acidification. Further research could also be applied to species which tolerate naturally high CO₂ (aq) levels, such as those inhabiting areas close to hydrothermal vents (Fabry *et al.*, 2008). If the underlying physiological and genetic mechanisms are isolated, it could enable shellfish industries to

produce ‘climate proof’ species in the future (Parker *et al.*, 2011).

Economically lucrative species, such as lobsters or crabs, are less likely to be affected through the direct impacts of ocean acidification, in comparison with marine bivalves. This is due to decapods’ ability to strongly regulate their internal ion concentration, which enables them to have a high level of biological control over ionic processes, e.g. calcification (Cooley and Doney, 2009), in addition to aiding the partial or full compensation of internal acid–base disruption, e.g. during hypercapnia induced acidosis (Spicer *et al.*, 2007; Small *et al.*, 2010). Currently, culturing production of crustaceans is negligible in the UK and Ireland. However, there have recently been successful developments to create lobster hatcheries, predominantly aimed at re-stocking natural populations (Burton, 2003). With this in mind, focus should be directed to potential energetic impediments in decapods as a result of maintaining ionic homeostasis under ocean acidification conditions.

Finfish are pre-adapted to some of the pressures brought about by increased ocean acidification (Ishimatsu *et al.*, 2004, 2008; Portner, 2008). To date, the majority of studies have not found any direct effects on growth development (Munday *et al.*, 2009a), egg survival (Munday *et al.*, 2009a; Franke and Clemmesen, 2011), metabolic activity (Melzner *et al.*, 2009) or swimming performance (Melzner *et al.*, 2009; Munday *et al.*, 2009a), indicating limited direct effects of ocean acidification for finfish farming in the near future. Indirectly, ocean acidification could affect farming through decreasing the olfactory responses of marine fish larvae, thus increasing their susceptibility to predators as they are less likely to be alerted to their presence (Munday *et al.*, 2009b, 2010). Further research is needed to understand the effects of ocean acidification on the calcareous structures in finfish, such as otoliths, stratoliths and gastroliths, to assess if there are any developmental impacts on movement or feeding throughout the life-cycle, which may affect both natural and farmed populations (Cooley and Doney, 2009).

In the short term ocean acidification is unlikely to be of major concern for the UK and Irish aquaculture industry. However, as a climate change phenomenon it has the potential to change entire food chains and community assemblages, and it is therefore necessary to further understand

the principal processes and effects of ocean acidification on ecosystems. Since the response to increased acidification is species specific, aquaculture research needs to focus on the commercially important species on a local scale. Further research on economically important shellfish is essential, with primary focus on larval development and spat.

CLIMATE CHANGE AND THE PHYSIOLOGY OF AQUACULTURE SPECIES

Most aquatic animals including fish are poikilotherms. Their metabolic rate and energy expenditure, and therefore their growth potential, are strongly affected by water temperature. Temperature increases that occur naturally during summer increase growth rates up to the tolerance limit of each species. Wild finfish avoid areas where the temperature is outside their natural temperature range, but farmed fish cannot, as they are confined to their cages.

Elevated water temperature will impact on fish directly through the influence of temperature on growth, and indirectly via specific nutritional and physiological processes that affect growth. Relationships between temperature and feed intake, growth, and growth efficiency have been broadly researched in the past for several salmon species and results show a bell shaped curve for the response of both feed intake and growth to increasing water temperature (Brett *et al.*, 1969, 1982). Several studies followed this classical approach to predict the optimum temperature for Atlantic salmon (Figure 4) (Koskella *et al.*, 1997; Larsson and Berglund, 2006). Feed intake is a

critical factor. Feeding to satiation means that fish will eat to meet their energy requirements, and feed intake typically increases rapidly with increasing temperature until appetite is inhibited at higher temperatures (Figure 4). Here, energy expenditure for maintenance increases and fish may not be able to cover their requirements for growth in addition to maintenance. As a consequence climate change may reduce growth in salmon in some areas due to a shift beyond the species' optimum temperature range.

Principal studies following a bioenergetic approach involving brown trout were conducted to describe energy requirements for growth (Elliott, 1976, 1982). Temperature was a key variable and its influence on feed intake, growth, body composition, nutrient retention efficiency and metabolic losses were modelled over a broad range of temperatures. Following these early studies, it was recognized that factorial approaches, which compartmentalise requirements to components, such as maintenance and growth, are essential tools in defining feed requirements in aquaculture (Bureau *et al.*, 2002; Lupatsch, 2009). Thus, under farming conditions, climate change related increases in temperature might even have a positive impact. It may result in enhanced growth and production, provided that feed supply satisfies the higher demand (Cho, 1992; Lupatsch, 2009).

Concepts and approaches developing growth models for salmon have been published (Iwama and Tautz, 1981; Jobling, 2003), but they need to be updated regularly, especially as genetic selection of farmed Atlantic salmon has improved the growth potential considerably. In Norway, for example, farmed Atlantic salmon have been selected for increased growth rate since 1975. Together with improvements in nutrition this has reduced the production cycle by approximately 1.5 years (Thodesen *et al.*, 1999; Gjedrem, 2010).

Generally, information about the impacts of higher temperatures on the life-cycle of Atlantic salmon is incomplete, since most studies focus on smaller, juvenile stages (Brett *et al.*, 1982; Elliott, 1991). There is little information about the impact of warmer waters on the nutritional requirements of large salmonids held in sea cages. Some understanding of nutritional physiology can be gained from the Tasmanian salmon farming industry (Carter *et al.*, 2005), as temperatures in those waters are on average higher than in the northern hemisphere. However, information on growth for salmonids reared in cages under

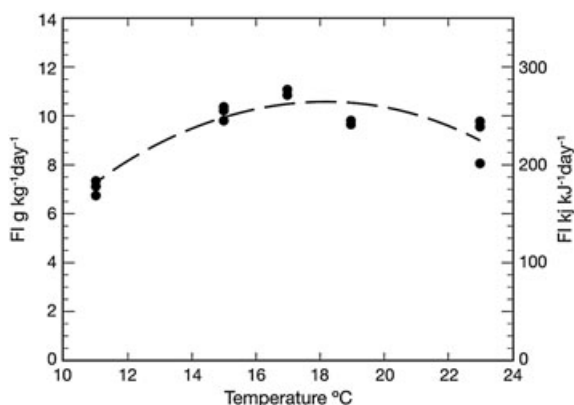


Figure 4. Predicted (dashed line) and observed (black dots) feed intake of age 1+ salmon (*Salmo salar*) reared at five different temperatures (after Koskella *et al.*, 1997).

farming conditions can rarely be found in the publically available literature. Currently, the best datasets on ration and growth, and probably the most sophisticated models, are those owned commercially by farms or feed companies.

Salmonids favour fairly low temperatures and the normal range for salmon farms lies within 5–19 °C. Fast, efficient growth is best achieved in water temperatures of 13–17 °C. Outside this range, production becomes less efficient, either due to slower growth or to temperature stress problems. Thus, a rise in water temperature of about 4 °C (IPCC, 2007) could exceed the optimal temperature range of salmon. In addition, seasonal extreme values may be equally or even more important than changes in annual average temperature. [Morgan et al. \(2001\)](#) found improved growth in rainbow trout during winter with a 2 °C temperature increase, but decreased growth in summer when the 2 °C increase was added to already high temperatures. Warming waters could open up new sites further north, or deeper waters, which are currently not suitable for aquaculture. However, areas may also become unsuitable. Currently salmon farming is concentrated in Scotland (Figure 2), where water temperatures range from 6–15 °C ([Baxter JM et al., 2011](#)). Increasing water temperatures may limit the southern expansion of salmon farms, although the main limitation for the foreseeable future is still likely to be the availability of suitable sites for cultivation.

A compounding factor is the decrease of water solubility of oxygen with increasing temperature. The worst combination is high water temperature and low water flow. This effect is more pronounced in sea water than in fresh water, as oxygen solubility is dependent on salinity. [Pörtner and Knust \(2007\)](#) suggested that the main physiological effects of climate change will be due to thermal limitation of oxygen. While many studies focus on juvenile life-history stages of salmonids, larger individuals might be at even greater risk, as they may reach their thermal aerobic limit sooner than smaller individuals ([Pörtner and Knust, 2007](#)). Thus, oxygen limitation could potentially affect larger fish in sea cages.

At present, most salmon hatcheries and broodstock facilities in Scotland rely on relatively simple flow-through systems with limited control of temperature and water quality. A combination of economic and environmental concerns is driving the industry towards the use of some land based recirculating systems which allow full control of

water temperature and quality. Impacts of climate change may necessitate that the majority of this production takes place in such systems to ensure the supply of good quality smolts, as high temperatures have been shown to detrimentally impact on egg quality and production ([King and Pankhurst, 2004](#)).

Opportunities for farming warmer water species such as sea bass (*Dicentrarchus labrax*) and turbot (*Scophthalmus maximus*) may emerge from rising seawater temperatures, as has been envisaged for Norwegian waters ([Bergh et al., 2007](#)). However, much of this farming is expected to be carried out in land-based recirculating aquaculture systems in the vicinity of the coast, as already widely practised in the Mediterranean for hatchery and nursery stages. Hence, issues such as the price of energy, land-use and market demand are likely to be of greater importance for the development of these new industries than changes in seawater temperature.

An increase in sea surface temperature is also likely to affect the growth of farmed shellfish. [Ferreira et al. \(2008\)](#) examined the potential effects of global climate change on shellfish growth by considering an increase in water temperature of 1 °C and 4 °C above the mean annual seawater temperature of Strangford Lough (N. Ireland, UK); the 1 °C increase scenario was based on near future predictions, and 4 °C is the maximum increase predicted for the year 2100 (IPCC, 2007). Using the ShellSim model of [Hawkins et al. \(2002\)](#), they predicted that an increase in water temperature would lead to a reduction in aquaculture productivity and a decrease in the mean weight and mean length of both oysters and mussels. These decreases would have a dramatic effect on the blue mussel, and lesser consequences for the Pacific oyster population. An increase of 1 °C in the water temperature would lead to a reduction of about 50% in mussel production and less than 8% in Pacific oyster production. An increase of 4 °C would result in a reduction of 70% in mussel production and less than 8% in Pacific oyster production.

HEALTH AND EPIDEMIOLOGY

Effects of climate change on the health of finfish and shellfish are currently not quantifiable. There are plenty of examples of emerging, or at least newly discovered, diseases in the marine or freshwater

environment (Lafferty *et al.*, 2004). However, the evidence that climate change is a major driver behind the emergence of diseases is thin (Lafferty, 2009). Cook *et al.* (1998) demonstrate a link between increasing sea temperature and spread of *Perkinsus marinus* in the eastern oyster *Crassostrea virginica*, as do Hofmann *et al.* (2001) for MSX disease *Haplosporidium nelsoni* in the same host species, although it is difficult to conclusively demonstrate causality in such observational studies. It has been suggested that global warming at best contributes to (viral) disease emergence and that translocation of hosts and vectors is the more important factor (Zell *et al.*, 2008).

Epidemic spread, a process driven by positive feedback, provides a noisy system which is relatively difficult to analyse robustly compared with other branches of biology. Surveillance of aquaculture diseases is further complicated by the large numbers of species involved for both hosts and parasites (Tables 3 and 4), and the comparative youth of the industry compared with terrestrial farming. The scarcity of long-term datasets hampers analysing and understanding the impacts of climate change on disease levels *in situ* (Karvonen *et al.*, 2010).

Fish parasites and pathogens

The potential impact of climate change on marine aquaculture in the UK and Ireland, in terms of health and epidemiology, depends on its impacts upon the key drivers for disease emergence (Figure 5). These drivers can be broadly grouped into parasite and microbial pathogen change and introduction, host change and introduction, and changes in contact and interaction between hosts and pathogens.

Parasites are subject to many of the environmental constraints experienced by other free-living organisms. However, they may also require extremely narrowly defined local environmental conditions, which are provided by a specific host, or several hosts, according to a given parasite's host specificity. At the same time the host environment may buffer parasites, particularly endoparasites, against wider environmental changes. Climate change will therefore affect parasites both directly through the ambient environment and indirectly via effects on host parameters such as distribution, behaviour, physiology, and mortality. The more complex the life-cycle in terms of the number of hosts and parasite stages involved, the more likely it

Table 3. Parasitic diseases in aquaculture species relevant to the UK and Ireland*

Aquaculture species	Parasite	Type	Comments	Key references
<i>Salmo salar</i> (Atlantic salmon)	<i>Lepeophtheirus salmonis</i> <i>Caligus elongatus</i> (sea lice)	Caligidae	Sea lice are a year round problem and are controlled through the administration of various bath and in-feed treatments. The efficiency of these may be reduced by altered environmental conditions, i.e. elevated water temperature, organic loading, reduced oxygen content, etc.	Johnson <i>et al.</i> (1993)
<i>Mytilus edulis</i> (blue mussel) and <i>M. galloprovincialis</i>	<i>Marteilia</i> spp.	Protozoan		Berthe <i>et al.</i> (2004)
<i>Ostrea edulis</i> (native oyster)	<i>Mytilicola</i> spp. (red worm) <i>Bonamia ostreae</i>	Copepod Haplosporidian		Bignell <i>et al.</i> (2008) Culloty and Mucalhy (2007)
<i>Crassostrea gigas</i> (Pacific oyster)	<i>Bonamia exitiosa</i> Various trematodes <i>Haplosporidium armoricum</i>	Trematoda Haplosporidian	Causes significant mortalities when introduced	Abollo <i>et al.</i> (2008)
<i>Homarus gammarus</i> (European lobster)	<i>Nicolthoë astaci</i>	Copepod	May be intermediate host	Renault <i>et al.</i> (2000) Hine <i>et al.</i> (2007) Wootton <i>et al.</i> (2011)

*This table shows main parasites affecting UK and Irish aquaculture species; it is not an exhaustive review of all diseases.

Table 4. Viral and bacterial diseases in aquaculture species relevant to the UK and Ireland*

Aquaculture species	Disease and agent	Type	Comments	Key references
<i>Salmo salar</i> (Atlantic salmon)	Amoebic gill disease (AGD) <i>Neoparamoeba pemaquidensis</i> Vibriosis <i>Vibrio splendidus</i> Herpes virus	Amoebae Bacterial Bacterial Viral	Reported from Scottish and Irish farms. Environmental changes predispose salmon to colonisation by amoebae and ciliates Mass mortalities usually during the summer Increasing importance in UK and Ireland as a major cause of disease.	Rodger and McArdle (1996) Bermingham and Mulcahy (2004) Anguiano-Beltrán <i>et al.</i> (2004) Le Roux <i>et al.</i> (2002) Segarra <i>et al.</i> (2010)
<i>Mytilus edulis</i> (blue mussel)	(Ostreid herpesvirus 1 (pvar))	Bacterial	Temperature dependent, more likely to affect larval stages and juveniles	Paillard (2004)
<i>Crassostrea gigas</i> (Pacific oyster)	Vibriosis (vibriosis) <i>Vibrio tapetis</i> Herpes virus Gaffkaemia (<i>Aerococcus viridians</i> var. <i>homari</i>) Shell disease syndrome	Bacterial Bacterial Viral Bacterial Bacterial (uncertain)	Brown ring disease in Manila clams No evidence of climate related changes in prevalence or severity/susceptibility Possibility of epizootic shell disease spread to European lobsters	Cawthorn (2011) Vogan <i>et al.</i> (2008)

*This table shows main viral and bacterial diseases affecting UK and Irish aquaculture species; it is not an exhaustive review of all diseases.

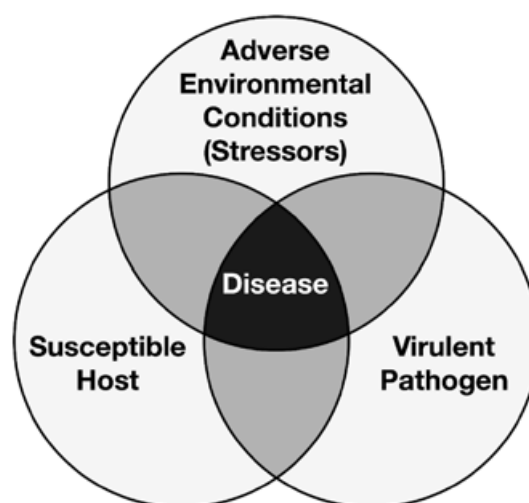


Figure 5. Disease susceptibility model for fish and shellfish.

is to be affected by environmental change, as each link in the life-cycle may be subject to different requirements (Overstreet, 1993). Effects on host parameters are relevant not only to cultured species but also to wild host reservoirs and vector species, which are common sources of infection for farmed animals.

There is evidence to show that specific environmental changes, whether 'naturally' occurring or anthropogenic, can have substantial impacts upon both protistan and metazoan parasites, affecting the identity of key parasite species within a host community, the dynamics of infection and the outcome of infection for the host (MacKenzie *et al.*, 1995; Williams and MacKenzie, 2003; Sures, 2004; Overstreet, 2007). While considerable work has been undertaken to predict the effects of climate change on parasitic diseases of terrestrial species, little work has been carried out to examine the potential effects of climate change on aquatic parasites (Marcogliese, 2001, 2008; Karvonen *et al.*, 2010). Moreover, the specialized, often highly simplified environment created by current aquaculture practices, particularly the high intensity monoculture systems frequently employed in the UK and Ireland, mean that predictions made for the impacts of environmental change upon parasitism of wild host populations may not hold true for cultured finfish, even within the same fish species.

Higher levels of parasitism in wild fish populations might be expected to increase host mortality, and thereby act to control parasite numbers. In contrast, parasites are controlled through interventions and

management strategies in aquaculture, rather than through parasite-associated mortality.

Effects of changes in temperature on parasites

Aquatic parasites show temperature optima for the successful completion of their life-cycle, and an extension of the season over which temperatures are amenable to survival and development will provide a longer window of infection for infectious stages. It will also provide more time for the proliferation of parasites to occur, both on and off hosts, and it will lead to more generations of parasites over the farm cycle through faster development times.

Higher temperatures may have direct implications for the level and nature of pathology experienced by hosts, where the speed of development of individual stages, the increase in numbers within the host or the final size and state of development reached are affected. With higher winter temperatures some parasites that would normally be dormant or present in negligible numbers, may continue to pose a problem over the winter months.

Since changes in water temperatures may affect movements of wild hosts, this may affect the introduction of parasites into aquaculture environments. The copepod parasite *Caligus elongatus* for instance, which principally infects marine cultured Scottish and Irish Atlantic salmon over summer and autumn, is brought into the aquaculture environment by wild marine fish species migrating into coastal waters to feed (Bron *et al.*, 1993; Revie *et al.*, 2002). The seasonal appearance and disappearance of this species from farm sites means that development of chemotherapeutant resistance is limited, in contrast to the salmon louse *Lepeophtheirus salmonis*, which is often present year-round. Longer residence times for migrating wild host species could therefore increase parasite incidence on farms, thereby affecting both fish health and parasite biology.

Increase in water temperature will also affect the ability to successfully treat parasite pathogens, with some chemotherapeutants being less suitable for treatment at higher temperatures. For example, the efficacy of hydrogen peroxide, used to remove caligid parasites from marine farmed Atlantic salmon, varies with temperature. It needs to be applied at lower concentrations for efficacy against lice at warmer temperatures, but it becomes

increasingly toxic to fish, with 7.7% mortality of Atlantic salmon reported at 14 °C and 100% mortality at 18 °C (Johnson *et al.*, 1993).

Effects of changes in water quality on parasites

Changes in precipitation patterns and the associated build-up of nutrients may lead to eutrophication and increased water turbidity. In farmed Atlantic salmon gill damage caused by elevated ammonium, nitrite and chlorophyll levels was suggested to be a predisposing factor for amoebic gill disease in Ireland, while temperature increase is itself a risk factor for the disease (Bermingham and Mulcahy, 2004). It has been suggested that the deteriorating quality of estuarine waters, combined with higher temperatures, may provide a risk in terms of numbers of the parasitic and toxic dinoflagellate *Pfiesteria* sp. (Burkholder and Glasgow, 1997; Rublee *et al.*, 2005), although this has yet to be reported as an issue for aquaculture. The species has nevertheless been documented in Northern European waters (Jakobsen *et al.*, 2002), suggesting that it could become a problem if coastal conditions change.

Deterioration in water quality or site hygiene will similarly increase ambient levels of many opportunistic pathogens (e.g. *Trichodina* sp.). The same will be true for bacterial pathogens, and opportunistic parasites may appear as secondary pathogens following bacterial infection, or they may themselves promote secondary infection by bacteria.

Changes in coastal salinity associated with changes in precipitation may also affect the survival and infection success of some marine parasites. The infective copepodid of *L. salmonis*, for instance, shows compromised survival at <29 ppt (Bricknell *et al.*, 2006) and it has been noted that sites with periods of high freshwater run-off experience loss of sea lice during low salinity flushing events (authors' observations).

Effects of changes in temperature on fish hosts

Changing climate may lead to some fish species extending their presence into new areas. These species will be exposed to the full spectrum of parasites present in the new environment. They may also carry their own parasites to add to the parasite load of native aquatic species, both farmed and wild. This could include harmful non-native species. The invading wild hosts might

act as host reservoirs for existing aquaculture pathogens.

Although parasites are introduced along with their exotic hosts, in general these hosts appear to be less infected than native species (Torchin *et al.*, 2003), possibly due to the stochastic extinction of small founder populations or the lack of local vectors.

With higher seawater temperatures the immune function of farmed fish species may increase within their optimal temperature range (Bly *et al.*, 1997; Bowden *et al.*, 2007). However, higher temperatures are associated with falls in oxygen levels, which is likely to increase stress in fish. This potentially compromises immune capacity and provides increased scope for opportunistic infections.

Parasite-host contact

Climate change may affect the pattern of contact between parasite and host populations, i.e. the routes of transmission. Changing physical factors may alter the spatial overlap between hosts and parasites. The contact could be accelerated or disrupted through altered ocean currents, changes in precipitation patterns or, in the long term, through sea-level change. One feature of climatic change that could affect UK and Irish aquaculture, and which was highlighted for the terrestrial environment by Hudson *et al.* (2006), is the possibility of increased 'spatial synchrony', i.e. the development of extensive areas sharing a similar climate or environment. It was suggested that under such conditions large-scale outbreaks of disease might follow through disease transmission across large areas. The timing of events in host and parasite life-cycles, such as hatching and migration, may shift to accommodate changes in the environment (Marcogliese, 2001). Because of the often complex interactions between parasites and hosts this may act to break or create conditions of spatial overlap, depending on circumstances, and levels of infection in wild or cultured hosts may increase or decrease.

While the above discussion principally deals with events occurring within the lifetime of hosts and parasites it is also critical to consider longer term evolutionary change. The genetics of both hosts and parasites have already been affected by culture environments and practices, for example breeding for desired farm traits such as disease resistance in salmon and the emergence of resistance to chemotherapeutant in bacteria and

sea lice (Treasurer *et al.*, 2000; Bravo *et al.*, 2008; Lees *et al.*, 2008a, b). The inevitable adaptations to climate change by hosts and parasites may result in unpredictable knock-on effects on resistance or virulence, depending on trade-offs between the traits under selection.

Microbial infections of fish

Climate change has a number of possible impacts on the infection of finfish by microbial and viral pathogens in aquaculture, both in terms of source and effects. Changes in rainfall and water temperature can influence the loading of pathogens within fish farms, as well as the species composition of those pathogens and the infection rates of the fish in the farm. Any increase in these infections would have a direct effect on the profitability of aquaculture.

Generally, in waters of the UK and Ireland pathogen infection rates are affected by water temperature in marine systems as a result of three factors:

- (i) Supply: bacteria and fungi both experience positive correlations between temperature and production. Therefore the reservoir of pathogens present in the environment is generally higher under warmer conditions, leading to increased infection risk.
- (ii) Active period: many diseases are only present in the population during warm, summer periods and are not active during the cooler winter months (Harvell *et al.*, 2002). Warmer waters will prolong the warmer periods and therefore extend the season during which fish will be susceptible to infection.
- (iii) Host susceptibility: climate change may shift conditions outside the biological optima of organisms. This is likely to place them under greater stress thus suppressing their immune system (Snieszko, 1974; Bowden *et al.*, 2007), resulting in an increased rate of microbial infection.

Reduced precipitation in the northern UK in summer (Lowe *et al.*, 2009) would reduce the frequency and loading of bacteria and fungi washed into the areas around finfish farms. However, for seasons where precipitation is predicted to increase and areas where rainfall events are going to intensify, the reverse will be the case, i.e. bacterial loading will increase and cultured animals will be more heavily affected by microbial pathogens. Species that are farmed near the border of their optimal biological niche

could experience stress and, hence, increased susceptibility to disease. This could be particularly problematic for cultured fish populations which lack the genetic diversity to counter new pathogens (FAO, 2011).

Several bacterial and viral infections of farmed fish in the UK are known to be influenced by temperature. Viral hemorrhagic septicemia virus (VHSV), infectious haematopoietic necrosis (IHN) and *Oncorhynchus masou* virus (OMV) are viral infections that show higher rates of infection at low water temperatures (8–15 °C) (Amend, 1970; Yoshimizu *et al.*, 2005; Arkush *et al.*, 2006). The risk of these diseases may decrease with increasing water temperature. Conversely, the bacterial infection enteric red mouth (ERM) becomes significantly more lethal at temperatures above about 10 °C, particularly when the host organism is stressed (Defra, 2011). Levels of furunculosis also increase at higher temperatures (Nordmo and Ramstad, 1999).

As has already been observed in larger marine organisms (Stebbing *et al.*, 2002; Nye *et al.*, 2009), increasing water temperature is likely to shift the range of pathogens northward. Until now some pathogens may have been unable to survive in UK and Irish waters, but are increasingly able to gain a foothold either through natural migration with the poleward spread of warm water species, or through the accidental introduction of organisms through stock movement (Birkbeck *et al.*, 2011). A documented example of the northward spread of a pathogen, although in shellfish rather than finfish, is that of the expansion of range of the Eastern Oyster Disease on the USA Eastern Seaboard in the 1980s (Ford, 1996). This expansion was linked to a series of relatively warm winters, which meant that the usual repression of the pathogen in the water during the cold months was not as effective as usual, and the infection was able to spread. Should a long-term change in water temperature take place, these expansions could become commonplace and also result in greater disease burdens in farmed finfish.

Shellfish

The form of shellfish aquaculture varies dramatically with different species and this will affect the likely changes in disease prevalence and severity following climate change. For example, mussels are either grown on ropes or seeded as mussel lays to be later harvested in offshore areas

around the coast and are therefore subject to the potential changes associated with climate change. At the opposite extreme, lobsters, although not currently produced commercially, have recently been subject to feasibility studies for production using land-based re-circulating aquaculture systems (RAS). These are not subject to primary environmental effects of climate change such as temperature change, acidification, and emerging diseases, because such systems have a high level of biosecurity and environmental control.

Many shellfish are susceptible to bacterial diseases such as vibriosis, especially during early stages of development (Table 4). Vibrios are ubiquitous in sea water and their growth is dependent on the temperature of the surrounding environment (Hsieh *et al.*, 2008). In simple terms, a rise in water temperature will usually result in higher bacterial growth leading to more disease. At the same time the host may be immune-compromised as a result of the increase in water temperature and associated changes in oxygen tension (Cheng *et al.*, 2003, 2004). Vibrios are often found living in association with the cuticle of crustaceans. For example, in the case of *Vibrio cholerae* a major environmental reservoir of such bacteria is a variety of zooplankton and phytoplankton (Colwell *et al.*, 2003; Pruzzo *et al.*, 2008). Therefore changes in plankton distribution as a result of climate change may affect the distribution of chitin-binding vibrios in seawater. Although there is little evidence of how climate change may affect the distribution of vibrios in the coasts around the UK and Ireland, in the warmer waters of the Mediterranean recent studies have shown a demonstrable link between *Vibrio* infections related to seasonality and temperature change (Vezzulli *et al.*, 2010).

Shellfish are also susceptible to rickettsia-like diseases. These bacteria are obligate intracellular parasites that can only multiply within the cells of their hosts. A recent study on a rickettsia-like disease, termed milky-disease, in the shore crab, *Carcinus maenas*, showed that increased water temperature during transport and storage and a general decay in environmental conditions (e.g. reduced oxygen solubility, nitrogenous waste products) resulted in high levels of disease that caused mortality (Eddy *et al.*, 2007). Hence, such diseases may also be influenced by changes in the temperature of surface waters and we would predict that the prevalence and severity of such

conditions may increase in temperate waters in the coming decades.

Several species of crustaceans are susceptible to shell disease syndrome – a condition thought to be associated with bacterial growth on the outer cuticle (Vogan *et al.*, 2008). In the USA, an emerging form of this disease, called epizootic shell disease, has caused major losses in lobster (*Homarus americanus*) populations. One of the factors that may be associated with this emerging disease is temperature, although other environmental factors may also be involved (Castro *et al.*, 2006; Glenn and Pugh, 2006).

The major parasitic diseases of shellfish are shown in Table 3. It is extremely difficult to predict how climate change may affect both the prevalence and severity of such diseases under different forms of aquaculture. In some cases, e.g. *Hematodinium* in crustaceans, the distribution of the host species may change the prevalence of disease, because there are other hosts or vectors as part of the parasite's life-cycle (Hamilton *et al.*, 2011). Hence, stocking an aquaculture facility with larval or juvenile animals may bring such diseases into the facility. The recent global emergence of dinoflagellate parasites within the genus *Hematodinium* (Morado, 2011) makes this a worrying condition both for fisheries and any planned development of crustacean aquaculture in the UK and Ireland.

There is some evidence that responses to climate change in terms of shellfish health and disease is species specific. Where species are on the edge of their boundaries and the prevailing conditions alter to more favourable conditions for diseases and parasites, or to more stress for the host, then epidemiologically the parasite or disease will have the upper hand and the host species could suffer high mortalities. Recent observations of *Mytilus edulis* and the red worm, *Mytilicola*, have shown that mussels high up the shore are exposed to higher temperatures and have higher levels of this parasite. They suffer high mortality compared with mussels lower down the shore. It is possible that increased temperatures would encourage higher *Mytilicola* numbers in animals further down the shore with economic consequences. However, where the conditions alter to be more favourable to the host, the disease or parasite is likely to be kept in check.

MACROALGAE

Currently the cultivation of algae in the UK and Ireland is negligible. There is, however,

tremendous interest in the expansion of the industry, primarily for the production of biofuel and for fish feed production (Spolaore *et al.*, 2006; BioMara, 2011), and hence climate change effects on algae need to be considered.

Macroalgae live in highly dynamic and hydrodynamically complex environments (Gaylord, 2000), experiencing forces that can often result in structural failure of the plants, reducing macroalgae cover over winter periods (Pratt and Johnson, 2002). These forces can be a major structuring driver of natural macroalgae communities (Menge, 1976; England *et al.*, 2008) and will be a determining factor on which species can be produced at any production site. Higher energy sites will be more suitable for kelp species such as *Alaria esculentus* and *Laminaria digitata* as opposed to *Saccharina latissima*, which favours less exposed situations (Lüning, 1990). If the frequency and strength of storms is going to increase there may be a shift to the production of species more tolerant of high energy environments. In addition to the increased risk of losing seaweed biomass through direct damage, change in storm patterns would also increase the hydrodynamic loading on the seaweed farm infrastructure, for example longlines and moorings. This could potentially lead to a further loss of the cultivated biomass through mechanical failure.

An alteration in nutrient loading of coastal waters due to increased extreme precipitation events will affect near-shore environments. Temporal variability in load can play a significant role in determining the response of macroalgae to nutrient enrichment (Worm and Sommer, 2000). The response to this temporal heterogeneity is species (Pedersen and Borum, 1996) and life stage (Bergstrom *et al.*, 2003) dependent. Increased variability in nutrient loading has been shown to benefit the dispersal, survival and growth of invasive algae (Incera *et al.*, 2009; Vermeij *et al.*, 2009), but may have little effect on slower growing kelp species (Pfister and Van Alstyne, 2003). It may retard the growth of some fucoids due to an increase of epiphyte growth (Pedersen and Borum, 1996). While there is generally little knowledge about the impacts of extreme precipitation events on kelp species, increased precipitation and increased temperature have been associated with a reduction of kelp cover in natural and artificial habitats of southern California (Grove *et al.*, 2002).

The effects of raising seawater temperatures as a result of climate change on macroalgal cultivation are not yet clear. It has been predicted that higher temperatures, coupled with lower wind velocities at certain latitudes, may result in a reduction of vertical mixing and a decreased supply of nutrients from deep waters, which in turn could lead to a drop in productivity (Bakun, 1990; Diffenbaugh *et al.*, 2004; Issar and Neori, 2010). As a consequence some areas may become unsuitable for the cultivation of macroalgae without the addition of extra nutrients. The boreal species *Ascophyllum nodosum*, for example, has its southern limit in the UK and the north of France. It has been in decline along the English coast close to Plymouth since the 1930s, but it flourishes in Brittany where strong tidal mixing reduces seawater temperatures and increases supplies of inorganic nutrients (Hiscock *et al.*, 2001). Similar supplies of nutrients are not present in the Plymouth area, and any changes in tidal mixing due to the impacts of climate change are likely to result in the decline of *A. nodosum*.

Many species of macroalgae are restricted geographically by the temperature ranges in which they will grow. There is some evidence of a shift in natural populations northward in the US due to increased seawater temperatures, particularly with regard to kelps (Graham *et al.*, 2007). For the UK and Ireland there are still many unknowns. Reports by the Environment and Heritage Service of Northern Ireland (2007) and Hiscock *et al.* (2001) suggested that there will be a decline in the distribution of some cold-water species of seaweeds as a result of climate change in Ireland, south-west England and Wales, with Scotland largely unaffected. Some of the cold water species found within the waters of the UK and Ireland are already at their geographical temperature limits, and if domesticated for a new aquaculture industry this industry is likely to be limited to regions where the effects of increased seawater temperature due to climate change are minimal (Environment and Heritage Service of Northern Ireland, 2007).

There has been considerable interest in the potential for macroalgae to sequester carbon on relatively short time scales (Gao and McKinley, 1994; Tang *et al.*, 2011; Chung *et al.*, 2011). Macroalgae in general contain approximately 30% dry matter, and at the current levels of cultivation the macroalgal industry removes nearly 0.7 million tons of carbon from the sea annually (Turan and

Neori, 2010). This represents only a very small impact on total carbon emissions, and for the offsets to be significant there would have to be a dramatic increase in the levels of cultivation (Turan and Neori, 2010). Potentially, a 1000 km² area of seaweed could sequester up to 1 million tons of CO₂ per year (Issar and Neori, 2010; Kraan, 2010). The impacts of increased atmospheric CO₂ on macroalgae cultivation will depend on the species that are being grown. Many macroalgae species, including commercial kelps, have CO₂ concentrating mechanisms (Axelsson *et al.*, 2000). These mechanisms ensure that the macroalgae are not carbon limited during normal growth. Growth rates of 13 species of macroalgae did not show any increase during experimental CO₂ enrichment (Israel and Hophy, 2002). However, there is evidence that the gross chemical composition of macroalgae may change (Zou and Gao, 2010) and this may be an important consideration for macroalgae cultivation, depending on the eventual utilization. As such, climate change will probably have a limited impact on the macroalgal cultivation industry of the UK and Ireland. The majority of impacts over the next 30 years are likely to come from changes in species composition of culture macroalgae and the need to develop engineering solutions; both as a result of increased storminess.

ABUNDANCE OF NUISANCE AND HARMFUL SPECIES

Climate change will potentially affect fundamental biological processes and would therefore influence the distribution, spread, abundance, and impact of invasive species (Gritti *et al.*, 2006), including harmful algal blooms (Hinder *et al.*, 2012). While not all invasive species will be nuisance species or cause harm, some taxa will exert substantial negative impact on native biota and economic values. Most marine aquaculture enterprises are vulnerable to nuisance and harmful species, which can have direct negative effects on the cultured species to the extent that they kill entire stocks within days.

Non-native species

Non-native species are those that have been intentionally or unintentionally introduced outside their native range (Maggs *et al.*, 2010). If these species become established and then threaten

biodiversity or cause economic damage, they are referred to as 'invasive' (Wilcove *et al.*, 1998). Climate change and invasive species present two of the greatest global threats to biodiversity and the provision of ecosystem services (Halpern *et al.*, 2008; Burgiel and Muir, 2010). There is growing evidence that in combination the threats have a compounding effect, particularly the increasing seawater temperatures and invasive species (Byers *et al.*, 2002; Stachowicz *et al.*, 2002). They can cause massive economic and ecological damage, and the estimated cost of non-native species to the economy in Great Britain is £1.7 billion a year (Vitousek *et al.*, 1997; Pimentel *et al.*, 2005; Williams *et al.*, 2010). The annual cost to the aquaculture industry in the UK and Ireland is estimated to be £7.1 million, although it was stressed that this is probably an underestimate as there is little distinction by the industry between native and non-native species during pest control operations (Williams *et al.*, 2010).

However, the damage to the UK and Irish aquaculture by non-native species, and the extent to which this may be attributed to climate change, is currently unknown. Widespread economic repercussions by non-native species to the UK aquaculture industry have not been reported to date, and there is little evidence at present to suggest that climate change is either exacerbating or suppressing the impacts that non-native species are currently having on the UK and Ireland aquaculture industry.

There are, however, examples from outside the UK and Ireland, which suggest that climate change could lead to an increase in the rate of invasions (Dukes and Mooney, 1999), particularly in marine habitats at higher latitudes (Reid *et al.*, 2009; Ruiz and Hewitt, 2009). The northward expansion of neo-tropical and temperate species has been seen across multiple taxa (Huntley *et al.*, 1995; Parmesan and Yohe, 2003; Hickling *et al.*, 2005; Sharma *et al.*, 2007; Ling and Johnson, 2009; Reid *et al.*, 2009).

Once a non-native species arrives in the UK or Ireland, its impact on the aquaculture industry could be extensive, either through the fouling of nets, cages, buoys, moorings, boat hulls and the cultured species themselves (Williams *et al.*, 2010), or through their competition for resources. There are several examples of non-native tunicates already present in the UK and Ireland, which have had a significant economic impact in other countries on aquaculture industries through

fouling: *Didemnum vexillum* (see Coutts and Forrest, 2007), *Ciona intestinalis* (see Ramsay *et al.*, 2008), *Styela clava* (see Thompson and MacNair, 2004) and *Botrylloides violaceus* (see Bock *et al.*, 2011). Stachowicz *et al.* (2002) found that *Botrylloides* sp. not only recruited earlier in years with warmer winters, but also grew at a faster rate than native tunicate species when the temperature was near the maximum observed in the summer months. They concluded that climate change may facilitate a shift in dominance of non-native species, some of which could prove costly for the aquaculture industry in the UK and Ireland.

Concerns for the aquaculture industry include the competition of non-native species for resources. This may be competition for space and food, or predation and parasitism by non-native species. The Pacific oyster *Crassostrea gigas*, for example, was introduced for aquaculture purposes in the 1960s, and has since colonized northern European Atlantic coasts. This has been linked to a decline in farmed oyster performance over the last 10 years in France (Cognie *et al.*, 2006). The range expansion coincided with increasing summer seawater temperatures in the region, and *C. gigas* can now be found in uncultivated regions on the south coast of England, north-east coast of Ireland, the French Atlantic coast, Norway and the Wadden Sea, where the dense intertidal hummocks of shell and live oysters can cause significant changes to the habitat structure (Figure 3b) (Nehls *et al.*, 2006; Syvret *et al.*, 2008; Dutertre *et al.*, 2010; Wrange *et al.*, 2010).

The distribution of non-native species is predicted to experience significant changes in response to climate change, with many already established species expected to spread northwards into new areas (Smith *et al.*, 2012). Improving early detection and long-term monitoring capabilities is urgently required, in conjunction with increasing research on the interaction between climate change and non-native species and the effect on maritime industries, such as aquaculture.

Jellyfish

The term 'jellyfish' describes various types of gelatinous zooplankton including scyphozoa, siphonophores and hydrozoa (Richardson *et al.*, 2009). Jellyfish tentacles contain toxic stinging cells or nematocysts which are used to immobilize planktonic and nektonic prey. Since jellyfish

numbers can also increase rapidly under the right conditions, blooms of jellyfish can cause serious problems for finfish held in sea cages (Purcell *et al.*, 2007).

Damage is caused by small jellyfish entering through cage meshes, or by tentacles of larger species breaking off and entering the cages. There is evidence that the major impacts of jellyfish to caged finfish come from damage to the gills (Baxter EJ *et al.*, 2011). A recent study demonstrated that even relatively short-term exposure (2 to 48 h) to macerated *Aurelia aurita* tissue could cause persistent gill damage to salmon (Baxter EJ *et al.*, 2011). Damage to the epidermis, necrosis of the gills (Bruno and Ellis, 1985), hypersensitivity to jellyfish toxin (Seaton, 1989) and secondary infections with pathogenic bacteria introduced via the jellyfish (Delannoy *et al.*, 2011) have all been implicated as causal agents in fish mortalities. Additional problems can be low oxygen caused by clogging of cage nets leading to stress in the fish which will affect food conversion and growth.

There are many examples of various jellyfish species causing fish kills, although they are often anecdotal and in some cases the species responsible was not identified (Purcell *et al.*, 2007). In August 1984, the leptomedusa *Phialella quadrata* was reported to have caused the death of 1500 Atlantic salmon smolts at an unnamed location in Shetland (Bruno and Ellis, 1985; Seaton, 1989). This appears to be the first documented case of salmonid mortality related to jellyfish in Scottish waters in the scientific literature. In November 2007, the small jellyfish *Pelagia noctiluca* swamped the cages of a salmon farm in Northern Ireland, killing an estimated 250 000 fish (Doyle *et al.*, 2008). Large blooms of this species have been reported many times, particularly in Ireland (Doyle *et al.*, 2008), and the implication is that jellyfish impacts on fish farms should be considered a real ever-present threat.

Different species of jellyfish are of concern. The lion's mane jellyfish (*Cyanea capillata*), the moon jellyfish (*Aurelia aurita*), the narcomedusa *Solmaris corona*, *Phialidium* sp., *Leuckartia raoctona* and *Catablema vesicarium* have all caused problems for aquaculture businesses (Seaton, 1989; Båmstedt *et al.*, 1998; Purcell *et al.*, 2007). Caged fish cannot escape a jellyfish bloom, and in some cases amelioration measures such as bubble curtains can make the situation worse if the jellyfish are broken up before they interact

with fish pens, and then the small fragments are able to pass through the net mesh into the cage.

Whether incidence of jellyfish induced fish kills are increasing due to climate change is equivocal. There is certainly growing concern over the negative impacts of jellyfish in marine systems. Central to this issue is the suggestion that overfishing may lead to a proliferation of jellyfish. Termed 'fishing down the food chain', it has been argued that the progressive removal of the largest fish in marine systems through overfishing, will eventually lead to an endpoint of marine ecosystems dominated by jellyfish and other invertebrates (for review see Richardson *et al.*, 2009).

In addition to the concerns of 'fishing down the food chain', climate change may also be affecting jellyfish abundance. There have been a number of studies in which time-series of jellyfish abundance spanning many years have been linked to various climate indices such as the North Atlantic Oscillation Index as well as water temperature and precipitation (Lynam *et al.*, 2004, 2011). Lynam *et al.* (2011) suggested that climate change was causing an increase in jellyfish abundance, and rising sea temperatures were cited as a factor (Graham, 2001; Mills, 2001). However, there are other potential causes for increases in jellyfish, including changes in the strength of influx of oceanic water to shelf sea areas (Lynam *et al.*, 2010), eutrophication (Arai, 2001) and even increases in hard surfaces, such as aquaculture rafts and dock walls, which can provide settlement sites for the benthic stages of some jellyfish species (Holst and Jarms, 2007; Lo *et al.*, 2008). In some regions, such as the Irish Sea, Japan, and Mediterranean, there is empirical evidence that warming conditions as part of climate change are leading to increased abundance of certain jellyfish species (Richardson *et al.*, 2009). However, in other areas, such as the North Sea, fewer jellyfish have been found in warmer years. Furthermore sometimes long-term trends in jellyfish abundance have not been sustained as time-series have lengthened, such as in the Bering Sea (Brodeur *et al.*, 2008). Owing to these varying long-term trends, generalizations about the impact of climate change on jellyfish abundance are hard to make (Richardson *et al.*, 2009).

In areas, such as the west coast of Ireland, where climate change is expected to cause an increased abundance of harmful jellyfish in the future, then increased fish kills at aquaculture farms are likely

to result. For example, the jellyfish *P. noctiluca*, which caused the fish kill in Northern Ireland, is generally considered a warm-water species that is periodically advected to the more northern latitudes of the UK and Ireland (Doyle *et al.*, 2008). For such warm water species we might expect that the progressive ocean warming of the last few decades in the NE Atlantic (Hobson *et al.*, 2008) will cause increased jellyfish occurrence at high latitudes. However, robust data to examine this prediction are lacking. Furthermore, current patterns may also be important in influencing the distribution of some jellyfish (Purcell *et al.*, 2007), and the interplay of ocean warming in addition to changing currents will need to be considered when assessing likely future trends in jellyfish abundance.

Jellyfish have been a component of the pelagic marine fauna that has generally been neglected by scientists, compared for example with crustacean zooplankton and fish. Hence there is a general lack of data on long-term trends in jellyfish abundance in UK and Irish waters. Jellyfish populations are naturally highly variable; many species respond to favourable conditions by rapid population increases, making them difficult to monitor (Boero *et al.*, 2008; Gibbons and Richardson, 2009). Observations of increases in jellyfish abundance over a limited time may only show a part of a cycle and may be followed by a subsequent decline (Brodeur *et al.*, 2008). These issues create problems for quantitative sampling of jellyfish populations, which are further exasperated by their patchy distribution. Hence large-scale efforts are usually required to produce robust time-series of jellyfish abundance (see Lynam *et al.*, 2011 for an example).

In Scotland individual farms and companies record incidents, but there has been little attempt to collate these data in order to identify regional or temporal patterns (Nickell *et al.*, 2010). There is therefore a clear requirement for improved monitoring to identify trends and to ascertain whether outbreaks occur in specific localities. Methods need to be developed that will allow routine estimation of abundance to reproducible standards at reasonable cost by farm operatives (Purcell, 2009). Promisingly, jellyfish surveys have been conducted using fisheries survey vessels that were simultaneously undertaking fish stock assessments, providing some of the best jellyfish time-series datasets to date in areas as diverse as the Gulf of Alaska, North Sea and Irish Sea (Lynam *et al.*, 2004, 2011; Brodeur *et al.*, 2008).

Continuation of existing surveys and initiation of new surveys to other areas and seasons is a central requirement to understanding the impact of climate change on jellyfish (see, for example, Bastian *et al.*, 2011). These surveys may also lead to an understanding of the physical–chemical conditions favoured by different species. In this way it may be possible to model the ‘climate envelope’ of conditions that species favour, and hence model both historical and future ranges, as has been done for other types of zooplankton where better survey data exists (e.g. copepods; Reygondeau and Beaugrand, 2011). The availability of such models could eventually provide warnings to the industry of impending blooms, allowing industry to take precautionary actions, such as harvesting or moving fish or employing physical barriers around cages. Development of new farm sites should also take account of the potential for jellyfish blooms. Current fish-farm husbandry practices need to be reviewed to ensure that they are not encouraging localized blooms by providing additional polyp substrate.

While jellyfish pose serious threats to aquaculture, in many Asian countries they are a resource for commercial harvesting (You *et al.*, 2007). Jellyfish may be cultured in ponds, often as part of a polyculture of other harvested taxa (e.g. sea cucumbers). In addition, jellyfish may be grown in artificial conditions and then ‘seeded’ into semi-enclosed bays where they grow in natural conditions prior to harvesting for human consumption. The global trend is for increased commercial harvesting of jellyfish, although such harvesting has yet to start in the UK or Ireland.

Harmful algal blooms

Harmful algal blooms (HABs) are a generic description for any algal growth that is considered by humans as being deleterious. Often such harmful or toxic bloom species are actually represented by a small fraction of the total algal (phytoplankton) biomass. Although typified by classic shellfish poisoning species, such as *Alexandrium*, many tens of genera have deleterious effects (Hinder *et al.*, 2011). Potentially any algal bloom, if large enough, can cause problems for aquaculture. In addition, most HAB species are not simply phototrophic phytoplankton, but are mixotrophic protists that can not only photosynthesize but also feed on dissolved and

particulate matter (including other plankton as well as detritus). As a consequence, their growth may be promoted by factors other than supply of inorganic nutrients. HABs are of particular concern for mussel cultivation and the rearing of caged fish.

Some HABs are more appropriately termed ecosystem disruptive algal blooms (EDABs) (Sunda *et al.*, 2006). These EDABs are formed by phytoplankton which is indigestible by zooplankton and hence block the normal flow of energy and elements through the food chain. Examples are large, mucus and foam-producing forms, such as *Phaeocystis*. Any prolific phytoplankton growth may potentially generate an EDAB, by becoming inedible (probably due to accumulation of noxious secondary metabolites) when they become nutrient starved (Mitra and Flynn, 2006). These organisms may also provide poor feed for bivalves, and when dying, EDABs cause further damage through decay (e.g. hypoxia). Here the term HAB will be taken to include EDABs.

The review of Bresnan *et al.* (2010) describes in some depth many UK-centric issues relating to climate change and HABs formation, and much of the information is relevant for the shellfish industry. HABs include dinoflagellates such as *Alexandrium*, associated with the production of paralytic shellfish poisoning (PSP), *Dinophysis* producing lipophilic shellfish toxins (LST), and diatoms such as *Pseudo-nitzschia* linked to amnesic shellfish poisoning (ASP). Incidences leading to closures of shellfish harvesting areas are reported from Scottish and Irish waters (Bresnan *et al.*, 2010).

The relationship between climate change and HABs is highly complex. Most HAB related events are near-shore, although they may originate off-shore through seeding and advection. In general, however, they are greatly affected by terrestrial processes, which are often linked to anthropogenic activity. For instance, nutrient inputs, changes to local marine ecosystems affected by land use and human population growth all have an impact on the occurrence of HABs. More significantly, increases in run-off, either via changes to catchment land use or by climatic drivers and climate change, also contribute to the occurrence of HABs. While eutrophication is a major driver in phytoplankton growth, an imbalance in N:P ratios within the organisms will promote toxicity (Youlian *et al.*, 1998). The emphasis on P removal in sewage treatment, coupled with the looming predicted

crisis in P-fertilizer supply beyond the next two decades, provides a high N:P supply ratio.

The ways in which climate change may affect N and P run-off is associated primarily with the pattern of rainfall, especially if flash floods overwhelm sewage treatment works. Changes in rainfall patterns may also have another impact associated with HABs, as a decrease in run-off may bring in less Si. Si stress is implicated with toxicity for the diatom *Pseudo-nitzschia australis*, now a common species off the UK (Youlian *et al.*, 1998).

However, marine processes equally affect phytoplankton growth and succession. The formation of algal blooms in coastal waters is associated primarily with sunny, warm, and stable water conditions. Elevated water temperatures enhance growth rates, creating a knock-on effect on zooplankton growth rates. The extent to which primary and secondary productions are enhanced pro rata, or whether HABs may proliferate, is not clear. In local situations where bivalves are important consumers of phytoplankton, factors affecting these and members of the meroplankton could be important. This may be especially so for the growth of mixotrophic species, which depend on the growth of other more benign organisms, such as the Diuretic Shellfish Poisoning (DSP) producing *Dinophysis*, which feeds on cryptomonads.

While terrestrial and marine processes have so far been the focal points of explaining HAB formation, ocean acidification is increasingly being recognized as a contributing factor. Algal bloom formation results in the alkalization of water, as CO₂, nitrate, and phosphate are consumed. The more eutrophic the water, the greater the biomass development (elevating pH), and the calmer the water conditions, the more rapid and higher the pH rises. This pH rise is detrimental to algal growth and is reported to affect species succession (Hansen, 2002). Through ocean acidification the starting pH for this process is lower, and hence the succession events may be expected to be altered. With the expected warming of the water, succession may be affected further. On collapse of the bloom the warmer waters and the lower final pH of the bulk water may also affect decay processes leading to changes in water quality beyond those experienced in recent centuries.

There is a long history of attempts to explain the apparent increased frequency in HABs on a global scale, with a corresponding continuing international research effort (GEOHAB, 2011). In

the UK, establishing whether such events are increasing has been hindered by uncoordinated and incomplete record keeping of plankton composition (Bresnan *et al.*, 2010). In addition, the form of the heavily indented western coastline can support locally important blooms which may go unnoticed. Unfortunately, remote sensing to detect algal blooms in UK and Irish waters is often problematic due to cloud cover, necessitating increased use of semi-automated samplers in near-shore waters to cover this gap in knowledge.

Bresnan *et al.* (2010) suggest that confidence in the UK science base is medium, but that our predictive ability is low. They identify a paucity of experimental and modelling studies of HABs species of UK importance as a problem and recommend enhancing our autecological understanding of the HAB species. In addition, we should improve our understanding of EDABs in general, while further investigating major UK and Ireland blooming phytoplankton. While the link to nutrient-status (water and phytoplankton), temperature and ocean acidification do need to be clarified, there is also a need for information on their competitors and consumers, both planktonic and benthic. Recent work has shown a profound shift in the relative abundance of diatoms and dinoflagellates in UK waters (Hinder *et al.*, 2012). Dinoflagellates (both HAB and non-HAB taxa) have declined markedly in abundance, while diatoms (both HAB and non-HAB taxa) have increased (Figure 6). These patterns have been linked to the combined effects of rising water temperatures and increasingly windy conditions (Hinder *et al.*, 2012).

EFFECTS ON GLOBAL FISHMEAL AND FISH OIL RESOURCES

Much of UK aquaculture, particularly salmonid, is currently dependent upon feeds formulated with relatively high levels of marine fishmeal (FM) and fish oil (FO) derived from industrial, feed-grade (reduction) fisheries of small pelagic species. The reduction fisheries are at their sustainable limits and, over the last 30 years, around 20–25 million tonnes (Mt) of feed fish have been caught annually, which reduce to about 6–7 Mt of FM and 1.0–1.4 Mt of FO (IFFO, 2011). Aquaculture is the fastest growing food sector with production increasing by an average of almost 9% per year (FAO, 2009). Between 1992 and 2006,

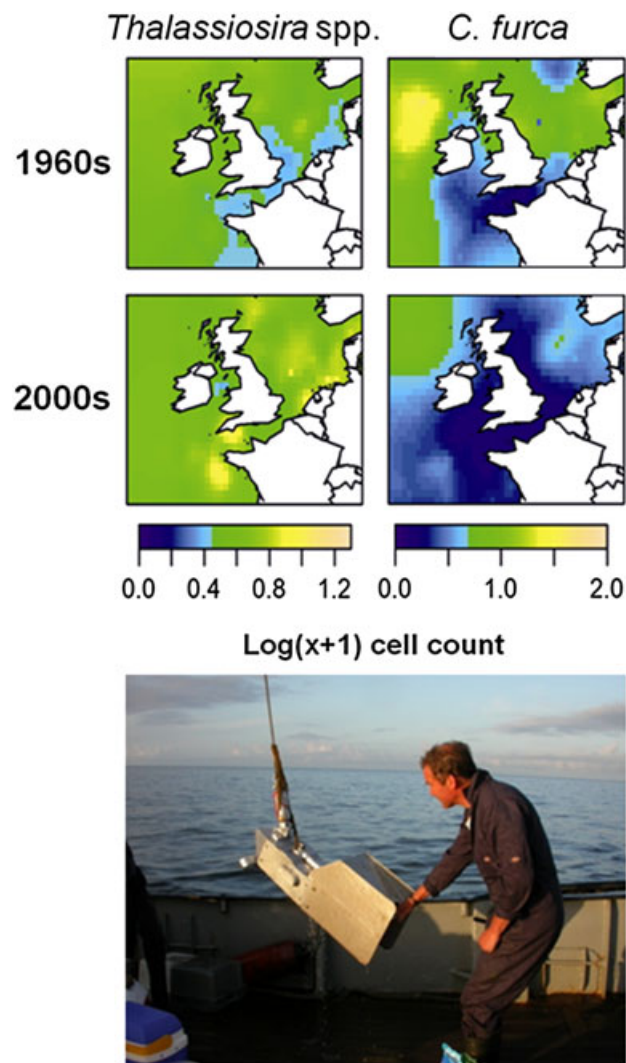


Figure 6. There have been profound changes in the abundance and distribution of dinoflagellates and diatoms in the NE Atlantic region (including the North Sea) related to increasing water temperatures and increasingly windy conditions. Dinoflagellates (both HAB and non-HAB taxa) have shown a marked decrease in abundance in Continuous Plankton Recorder (CPR) samples while diatoms (both HAB and non-HAB taxa) have tended to increase, so that the relative abundance of diatoms to dinoflagellates has increased. The plots show changes in the distribution and abundance of one diatom taxa (*Thalassiosira*) and one dinoflagellate (*Ceratium furca*) derived from the CPR survey (modified from Hinder *et al.*, 2012). Colour scale is based on log transformed numbers of cells per CPR sample and provides a relative measure of the abundance of taxa. Image shows a CPR being deployed in the North Sea.

consumption of FM and FO for aquaculture increased from around 15% and 20% of global supplies, respectively, to around 68% and almost 89%, respectively (Tacon, 2005; Tacon and Metian, 2008). There is no realistic prospect of FM and FO production being increased in the future, and indeed, there is increasing competition for these small pelagic species for direct human consumption (Tacon and Metian, 2009a, b). Therefore, the global FM and FO market is one

of finite and strictly limited supply and ever-rising demand. It is accepted that climate change is likely to play a significant role in changing the productivity, seasonality and distribution of marine ecosystems and their resources in the future (Glantz, 2005). For instance, there are increased captures of boarfish (*Capros aper*) in the NE Atlantic off Ireland that may be related to climate change affecting stocks. This species is contributing to FM (although not FO) production, but they may simply be displacing other pelagic species, mainly sandeel, rather than representing an increased resource. Therefore, looking forward, global FM and FO supplies are uncertain and currently our capacity to predict and respond to these impacts is limited (Merino *et al.*, 2010a, b).

The largest reduction fishery is on the eastern South Pacific seaboard (Chile/Peru; anchovies and sardines), with other major fisheries in Asia (Thailand and Japan), North America (USA, Mexico and Canada; menhaden, capelin) and Scandinavia (blue whiting, herring, mackerel, etc.). Climatic events are well-known to affect FM and FO supplies. The El Niño-Southern Oscillation (ENSO), which occurs periodically across the tropical Pacific Ocean, is characterized by variations in surface ocean temperature (El Niño, warming and La Niña, cooling), which can dramatically affect the reduction fisheries. El Niño reduces the upwelling of cold, nutrient-rich water decreasing the phytoplankton that sustains the large fish populations, and in the 1972 event the Peruvian anchoveta reduction fishery collapsed. The last major El Niño in 1997–1998 reduced production of FO to under 0.8 Mt and of FM to around 5 Mt (IFFO, 2011), which affected aquaculture through increased prices. If repeated now, a similar reduction in global FO stocks would have severe supply implications as well. Climate change impacts are likely to amplify these natural variations and to exacerbate existing stresses on marine fish stocks. Studies of global warming in the ocean have shown that the south-east Pacific region has not registered a sea surface temperature increase during the last 50 years, as has happened in most other regions. However, Peru has initiated a major study to assess the impact of climate change on the sardine and anchovy fisheries in the Northern Humboldt current system, the Peru Ecosystem Projection Scenarios project (PEPS, 2009). Effects of climate change on other fisheries are under similar

investigation, and the UK project, QUEST-Fish, is currently investigating how climate change would potentially affect future production of global fisheries resources (Barange *et al.*, 2010a, b). Although there is a slight but definite downward trend in global captures of feed fish in the last 30 years (IFFO, 2011), there is no direct evidence that this is solely due to climate change, and other factors, not least fishing pressure, could be the cause.

Within the aquaculture sector, the continued use of FM and FO as major feed ingredients has been of considerable concern for many years. The finite, limited supply and rising demand for global FM and FO has resulted in generally increasing prices with climatic variability introducing further instability and pressure on supply and prices (Mullon *et al.*, 2009). There are also increasing concerns over the use of marine products for animal feeds, with aquaculture especially perceived to be driving exploitation of marine fisheries (Naylor *et al.*, 2000). Regulation of the permitted levels of contaminants in animal feeds is also a concern restricting the use of FM and, especially, FO from specific geographical locations such as the Baltic (Tocher, 2009). All these factors have dictated that, for aquaculture to continue to expand, solutions to the problems of FM and FO supply must be found and this has prompted considerable research (Naylor *et al.*, 2009). Most of this effort has been directed towards developing other, more sustainable alternatives as replacements for FM and FO as feed ingredients (Tocher, 2009). Alternative marine resources such as exploiting lower trophic levels (zooplankton) to produce meals and oils, and/or the use of fisheries by-products (white fish and aquaculture wastes) have significant ecological, health and other worries (Olsen *et al.*, 2010). Therefore, the current favoured solution is the use of terrestrial plant meals and vegetable oils as replacements for FM and FO, respectively. Maximal levels of replacement that can be applied without affecting production and nutritional quality vary with species, with salmonids allowing greater levels of replacement than marine species such as cod, sea bream, and sea bass. Levels of replacement also vary with region, with replacement of FM and FO in Atlantic salmon farming, being higher in Chile and Norway than in the UK. These efforts along with improvements in farming practices have

greatly increased the efficiency of aquaculture activities in converting dietary nutrients into human food, often calculated as a fish-in fish-out (FIFO) ratio (Kaushik and Troell, 2010). Thus, the FIFO ratio is decreasing for all species across all aquaculture sectors (Tacon and Metian, 2009a, b).

The FM/ FO debate has also focused attention on the great differences within global aquaculture, with the greatest use of FM and especially FO for the high-value, high-trophic level carnivorous species generally farmed in the economically developed countries (Tacon *et al.*, 2010). This has prompted the suggestion that future aquaculture development should focus more on farming of species of lower trophic level and less dependent upon dietary FM and FO. This would require some change in consumer perception and acceptance of new, less familiar, farmed species. A similar situation exists with another possible long-term solution such as the use of transgenic approaches, primarily the engineering of plant crops to suit aquaculture needs, but also the possibility of engineering farmed fish to suit the alternative feeds (Tocher, 2009).

CONCLUSION

Direct effects of climate change on the UK and Ireland aquaculture sector have so far not been quantifiable, and current evidence of impact is speculative. There are several potential reasons for this.

First, it is possible that climate change has indeed so far not translated into measurable effects on the UK and Ireland aquaculture sector. Impacts such as storms, diseases or temperature variations are continuously dealt with by the industry, and it is difficult to separate climate change from natural variations in environmental conditions. Also, the aquaculture sector is relatively young in the UK and Ireland and the pace of technological progress has been rapid, as illustrated by developments in cage design, feed formulation, and pharmaceuticals. Swift adaptations and high resilience of the industry may have outweighed the comparatively slow pace of climate change thus far. Although future problems with parasites and diseases in finfish aquaculture may be controlled through intervention, it will be at increasing cost.

Importantly, measuring the effects of climate change demands systematically collected monitoring

data. This is a pre-condition for establishing baselines and discovering significant trends against which new observations can be compared. While such long-term data exists for wild fish populations from stock assessments, information is sparse for the aquaculture sector. Some relevant data on, for example, storm damage, growth rates, availability of spat or fouling of cages, is held by individual enterprises, but little of this is publicly available, nor is it collated systematically.

The level of threat from climate change depends on the degree of reliance that individual sections of the aquaculture industry place on prevailing environmental conditions. Bivalve farming, which depends on wild spat for stock, plankton for food and water quality for health, is highly susceptible to various effects of climate change. In comparison salmon farming is more independent from the natural environment since feed supply and offspring are managed. Nonetheless, cages are vulnerable to storm damage and fouling, and the fish are exposed to parasites and pathogens occurring in the surrounding water. At the other end of the spectrum, marine aquaculture systems can be engineered to completely isolate livestock from the surrounding environment, e.g. land-based recirculating aquaculture systems. It is anticipated that this form of aquaculture will increase in the future to enable continued supplies of established aquaculture species, irrespective of changing environmental conditions.

In terms of opportunities arising from warming coastal waters, it is compelling to envisage the possibilities of rearing warmer water species. For the bivalve industry climate change could open opportunities for rearing the Pacific oyster in waters currently unsuitable for the species, although in the wild this non-native species is likely to compete with mussels or native oysters. Similarly, warmer water finfish species, e.g. sea bass, could potentially displace salmon farming in certain locations. While this is technically a valid argument, market forces are more likely to determine whether or not a new species is cultured. Salmon production meets consumer demand; the industry is established and profitable. If environmental conditions became unsuitable for salmon cultivation, it is considered more likely that measures will be taken to overcome the emerging obstacles, rather than endeavouring to substitute the species.

However, demands do change. New opportunities may arise by shifts and changes in wild fish stocks in

response to climate change. If, for example, cod landings were to decline substantially while the consumer demand remained, higher prices would open opportunities to expand the rearing of this species. Further opportunities may arise from hatching early development stages of shellfish and finfish for the restocking of wild populations. Small-scale experiments are already underway for lobsters and bivalves, but this process could gain importance for other species. Hence, the effects of climate change on wild fish stocks and fisheries may indirectly have a greater effect on the UK and Ireland aquaculture sector than direct effects on currently farmed species.

Invading species and particularly jellyfish pose a real threat because they can wipe out entire salmon cage sites within days. Since jellyfish blooms are suspected to be affected by climate change, and their abundance is possibly linked to developments in the wild fisheries, efforts to improve the monitoring of jellyfish should be supported.

Possibly the greatest challenge for marine fish farming lies in the replacement of fishmeal and fish oil as a main ingredient in formulated feeds. Climate change plays a part in the declining stocks of small pelagic fish species such as anchovies and sardines, although fishing pressure is likely to be of greater importance. Furthermore, fishmeal and fish oil is a finite resource and limits the expansion of the aquaculture sector. Terrestrial plant meal and vegetable oils are currently being evaluated as replacements, but their production competes with other farming and land-use interests. An opportunity for the UK and Irish aquaculture sector arises from the production of omega-3 rich oils from microalgae and other microorganisms.

Improving understanding about the degree to which climate change affects aquaculture in the UK and Ireland requires research that addresses large spatial and temporal scales. In order to gain scientifically rigorous insight into effects of climate change on the aquaculture sector, publically available nationwide information needs to be collected, at least for the most pressing threats to the industry. However, while long-term monitoring is desirable, this may be unlikely to be resourced in the current and foreseeable economic climate. There may have to be reliance upon an iterative development of predictive models, which are validated using a patchwork of data emerging from experimental and observational studies.

In the near term, priority should be given to elucidating the impact of climate change on bivalve aquaculture, as this sector has fewer technological and pharmaceutical tools to respond to challenges. In addition, the effects of changing rainfall and run-off patterns on coastal water quality need clarification. This is particularly important in mussel growing areas that are located in shallow waters and estuaries, such as is common in England and Wales. Understanding the likelihood of deteriorating water quality due to climate change is an urgent challenge.

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