Food Research International 43 (2010) 1766–1779

Contents lists available at ScienceDirect

Food Research International

journal homepage: www.elsevier.com/locate/foodres

Climate change and seafood safety: Human health implications

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ARTICLE INFO

Article history:
Received 11 November 2009
Accepted 4 February 2010

Keywords:
Climate change
Seafood
Harmful algal blooms
Pathogens
Trace metals
Organic chemicals

ABSTRACT

Worldwide, anthropogenic climate change is now a reality and is already affecting the biology and ecology of some organisms, as well as several chemical pathways. Little is known about the consequences of climate change for the food system, particularly seafood, comprising all stages from "farm to fork" (mainly primary production, processing, transport and trading). In this context, the current review aims to elucidate climate change impacts on seafood safety and its human health implications. Both chemical and biological risks are foreseen to impair seafood safety in the future as a consequence of climate change; in particular, toxic metals, organic chemicals residues, algal toxins and pathogens of both humans and marine organisms. However, different species respond differently to such stresses. Public health authorities will face new challenges to guarantee seafood safety and to sustain consumers' confidence in eating seafood in a warmer world.

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1. Introduction

Anthropogenic climate change is one of the greatest environmental challenges the world faces today. Over the past two centuries, human activities have resulted in dramatic increases in atmospheric greenhouse gas emissions, including carbon dioxide (concentrations of CO_2 have increased by more than 30%, from 280 ppm in pre-industrial times to present day levels of 387 ppm (Earth Systems Research Laboratory/National Oceanic and Atmospheric Administration; http://co2now.org/)). Other greenhouse gases that have undergone considerable increases are methane, nitrous oxide and aerosols like chlorofluorocarbons (CFCs) (Forster et al., 2007). These molecules play a key role in global warming by absorbing infrared radiation and trapping heat near the Earth's surface. This has already and will continue to alter the planet's climate in several ways that can ultimately affect human health (Confalonieri et al., 2007). Climate change impacts include warmer air and seawater temperatures, increased ocean acidity, increased sea level, and changes to the intensity of weather disturbances, precipitation and wind patterns (Solomon et al., 2007). These changes have decreased the quality, quantity and safety of food (Easterling et al., 2007), degraded land, and decreased biodiversity and ecosystem function (Fig. 1).

Climate change impacts are expected to worsen over the next decades (Solomon et al., 2007). Model results indicate that even if greenhouse gas emissions are capped at present day levels, some warming will still occur because of the carbon dioxide that has already accumulated in the atmosphere. The ensemble of models used by the United Nations Intergovernmental Panel on Climate Change (Solomon et al., 2007) estimate that by the end of the 21st century: (a) atmospheric carbon dioxide concentrations will be as high as 730–1020 ppm; (b) global mean surface temperature will increase by 1.1–6.4 °C; (c) sea-level will rise by 0.18–0.59 m due to thermal warming and melting glaciers and ice sheets; (d) the pH of seawater will decrease by 0.14–0.35 units; (e) the Atlantic Ocean Meridional Overturning Circulation may decrease by up to 50%; and (f) the water cycle will accelerate, with increased precipitation in tropics and high latitudes, drier conditions in subtropics, and increased frequencies of extreme droughts and floods. The Earth has not experienced variations of this magnitude on this short a timescale in the past several million years (e.g., Field, Baumgartner, Charles, Ferreira-Bartrina, & Ohman, 2006; Petit et al., 1999), and the consequences to future generations are largely unknown (Solomon et al., 2007).

The direct impacts of climate change on various aspects of human and animal health and welfare is a topic widely debated. However, the indirect consequences of climate change for the food system, particularly seafood, comprising all stages from "farm to fork" (mainly primary production, processing, transport and trading), have received less attention. In this context, the current review discusses the potential effects of climate change on seafood

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doi:10.1016/j.foodres.2010.02.010

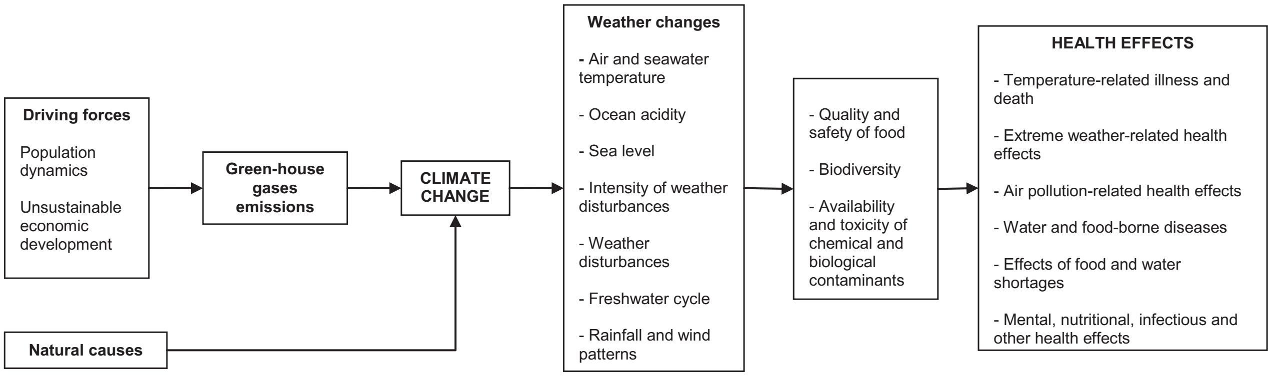

Fig. 1. Consequences of major climate change to human health.

chemical and biological safety and the implications for human health.

2. Discussion

2.1. Climate change and the oceans

About 520 million people (i.e., around 8% of the world's population) depend on seafood as a source of protein, income or family stability (FAO, 2009). The oceans and estuaries (i.e., productive coastal environments where rivers meet the sea) that support these fisheries are projected to experience numerous changes as a result of anthropogenic climate change. Some of the most robust projections include warming of the upper ocean, an acceleration of the water cycle which may change the timing and volume of freshwater entering the coastal ocean, and increased stratification of the ocean (Hansen et al., 2006; Meehl et al., 2007). Increased stratification is predicted to occur from warming and freshening of the surface layer of the ocean, which increases the stability of the water column and inhibits vertical mixing (Solomon et al., 2007). This can reduce the amount of nutrients that are mixed into the surface waters from deeper underlying waters, with consequences for the growth of phytoplankton and other marine organisms that occupy higher trophic levels in the food chain. Stratification and nutrient concentrations at coastal boundaries may also vary due to changes in upwelling patterns. Upwelling is a wind-driven process that causes cold, deep and nutrient-rich water to be uplifted at the coast (Smith, 1968). This natural injection of nutrients typically occurs along western boundary currents of the world's oceans and it supports productive fisheries in these regions (e.g., the Peruvian anchovy fishery) (Malakoff, 1998). This process has also been demonstrated to strongly influence toxin-producing harmful algal blooms that can contaminate seafood (Malakoff, 1998). The influence of climate change on upwelling patterns is less certain, and in some cases future projections of upwelling winds are conflicting (e.g., Fraga & Bakun, 1993; Álvarez-Salgado et al., 2008). Other ways that climate change may impact the ocean include changes to circulation patterns, the frequency of extreme events such as hurricanes, sea-level rise, and ocean acidification resulting from changes to seawater carbon chemistry (Holgate, 2007). These changes may alter the concentrations of chemical and biological contaminants of seafood. The remainder of this review will discuss these potential changes and speculate on the consequences for seafood safety and human health.

2.2. Climate and chemical contaminants of seafood

Human activities have resulted in the release of several chemical contaminants into the environment in the last decades. These include persistent natural chemicals, like polycyclic aromatic hydrocarbons (PAHs), toxic metals (e.g., mercury, cadmium, lead, zinc and copper), and synthetic organic chemicals, either produced specifically for industrial or agricultural purposes or arising as by-products from combustion or industrial processes (e.g. dioxins, tributyltin (TBT), polychlorinated biphenyls (PCBs)) (Schiedek, Sundelin, Readman, & Macdonald, 2007). Chemical contaminants enter marine ecosystems via direct discharges from land-based sources (e.g., industrial and municipal wastes), river runoff or drainage, atmospheric deposition from local and distant sources, and ships (Schiedek et al., 2007). Many contaminants accumulate in sediments, where they can remain for very long periods, and in the food-web where they can reach high concentrations in top-level predators and ultimately affect human health.

Climate change impacts on hydrographic conditions are expected to directly influence the availability and toxicological effects of chemical and biological contaminants (HELCOM, 2006; Schiedek et al., 2007; Fig. 2). Warmer water temperatures and changes to precipitation and stream flow patterns may exacerbate many forms of water pollution, including sediments, nutrients, dissolved organic carbon, pathogens, pesticides, toxic metals and salts (Boorman, 2003; Kundzewicz et al., 2007; SWCS, 2003). The increase in bioavailability of chemical contaminants and changes in metabolic rates and enzyme activities of marine organisms may influence the metabolism and detoxification of toxic substances (HELCOM, 2006). Consequently, altered prevailing environmental conditions would impact not only the uptake, retention, and detoxification of chemical contaminants, but also the exposure of marine organisms to toxic substances. Studies show that several plant and animal species are more vulnerable to chemical contaminants in warmer temperatures (Noyes et al., 2009). It is not clear if this relationship will dominate in the complex world of multiple stressors, but it does support the argument to minimize all exposures to chemical contaminants as we seek to meet the challenges of a warming world.

2.2.1. Toxic metals

Toxic metals are considered to be hazardous when present above threshold concentrations. This threshold concentration depends on the metal, animal species and environment. Particularly, cadmium, lead and mercury are regarded as the most toxic metals in seafood products, but methyl mercury is the most dangerous. Although most marine organisms tend to accumulate and use for metabolic purposes toxic metals from the environment, they are also capable to store, excrete (through faeces, eggs or moulting) or detoxify (binding to proteins such as metallothioneins or to insoluble metaliferous granules) many of them (reviewed by Rainbow, 2002). The main mechanisms of toxicity of toxic metals are related to the osmotic disturbances and alterations of enzyme syn-

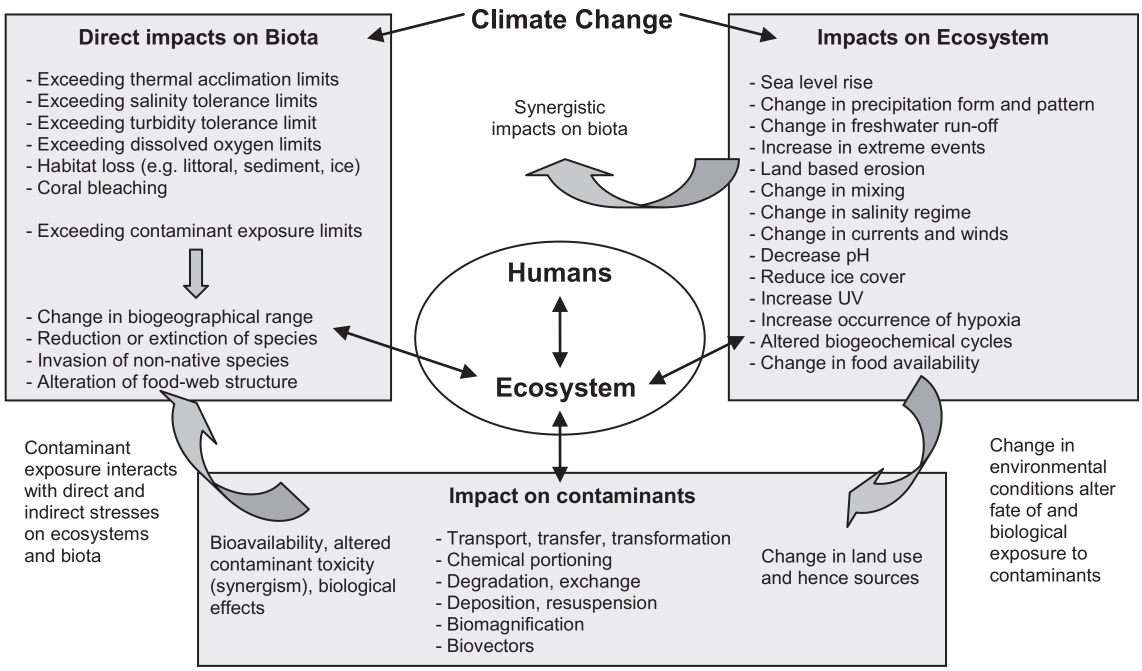

Fig. 2. Overview of climate change impacts on ecosystem and biota, and how they interact with contaminants, and their fate and effects (adapted from Schiedek et al. (2007)).

thesis and activity (Jezierska, Lugowska, & Witeska, 2009). Additionally, toxic metals in seafood can also affect various physiological processes in these organisms, including tissue damages, inability to regenerate damaged tissues, growth inhibition, damages to genetic material such as DNA, and changes in breeding and development (Jezierska et al., 2009). The initial period of embryonic development, just after fertilization, and probably the period of hatching are the most sensitive to metal intoxication, resulting in disturbances of developmental processes and causes embryonic and larval malformation and mortality (Jezierska et al., 2009). The changes induced by metals in marine organisms' embryonic development may be related to intoxication of spawners, accumulation of metals in eggs and spermatozoa, or a direct effect of metals on the fertilization process and on the development of embryos due to the entrance of metals through the egg shell. Such physiological changes in marine organisms can be amplified by changes in environmental conditions (Jezierska et al., 2009).

The salinity of coastal and estuarine systems will experience fluctuations arising from changes to precipitation and stream flow patterns. Salinity may affect the toxicity of various classes of toxic metals due to either bioavailability or physiological factors. In particular, metals like cadmium, chromium, copper, mercury, nickel and zinc are taken up more rapidly by phytoplankton/fungi, annelids, bacteria, molluscs and crustaceans at reduced salinities (Hall & Anderson, 1995). This is because of the action of fresher water on osmoregulatory mechanisms, which likely increases the bioavailability of the free toxic metal ions (Blackmore & Wang, 2003; Hall & Anderson, 1995; Ni, Chan, & Wang, 2005; Roast, Rainbow, Smith, Nimmo, & Jones, 2002; Zhang & Wang, 2007) (Fig. 3). Animal physiology plays also an important role in the toxicity of various metals at a range of salinities, as marine and euryhaline species respond differently to changes in salinity during toxicity tests. Euryhaline species are most resistant to toxic conditions at isomotic salinities due to minimization of osmotic stress (Hall & Anderson, 1995). In contrast, arsenic toxicity is not affected by salinity, despite the most toxic form arsenite (3+) predominating over arsenate (5+) at low salinities (Fig. 3; Bryant, Newbery, McLusky, & Campbell, 1985a). However, the authors of this study measured the As anionic form instead of the cationic one, and so it is likely that the method used to measure the anionic form may have affected the As bioavailability and also influenced osmoregulation. There is still a shortage of studies concerning the influence of salinity to on the toxicity of lead, silver and selenium (Fig. 3). Several authors reported increased toxicity of cadmium to the estuarine crab *Paragrapsus gaimardii* (Sullivan, 1977) at lower salinities (8.6–34.6 ppt) and of mercury to the mangrove clam *Polymesoda erosa* at higher salinities (10–30 ppt) (Modassir, 2000). The decrease of mercury toxicity at lower salinities could be due to the enhanced ability of the animal to eliminate this toxic metal from the body (e.g., increased water flow, ventilation rates) or due to changes in the mercury chemical form and chemical interaction of the metal in seawater, thus affecting bioavailability (Modassir, 2000). Bryant, Newbery, McLusky, and Campbell (1985b) tested the acute toxicity of nickel and zinc on two estuarine invertebrates (*Corophium volutator* and *Macoma balthica*) at a wide range of salinities (5–35 ppt), and found that survival of these species decreased in the presence of both metals as salinity decreased.

Temperature-related increases in the uptake, bioaccumulation and toxicity of metals have been reported for several marine organisms, including crustaceans, echinoderms and molluscs (Hutchins, Teyssii, Boisson, Fowler, & Fisher, 1996; Sullivan, 1977; Wang, Chuang, & Wang, 2005) (see Table 1). However, different species respond differently to such stresses. Warmer water temperatures facilitates mercury methylation, and the subsequent uptake of methyl mercury by fish and mammals has been found to increase by 3–5% for each 1 °C rise in water temperature (Booth & Zeller, 2005). The uptake of cadmium and zinc was found to be temperature dependent over a range of 15–30 °C in green mussel *Perna viridis* (Wang et al., 2005). The blue mussel *Mytilus edulis* when acclimated at 2 and 12 °C was found to accumulate up to 5.7-fold more silver, 5.3-fold more americium, and 2-fold more zinc at lower temperatures, largely because these metals were assimilated from food more effectively at lower temperatures (Baines & Fisher, 2008). In contrast, *M. edulis* bioaccumulation of aqueous and dietary cadmium and cobalt was higher at 12 °C com-

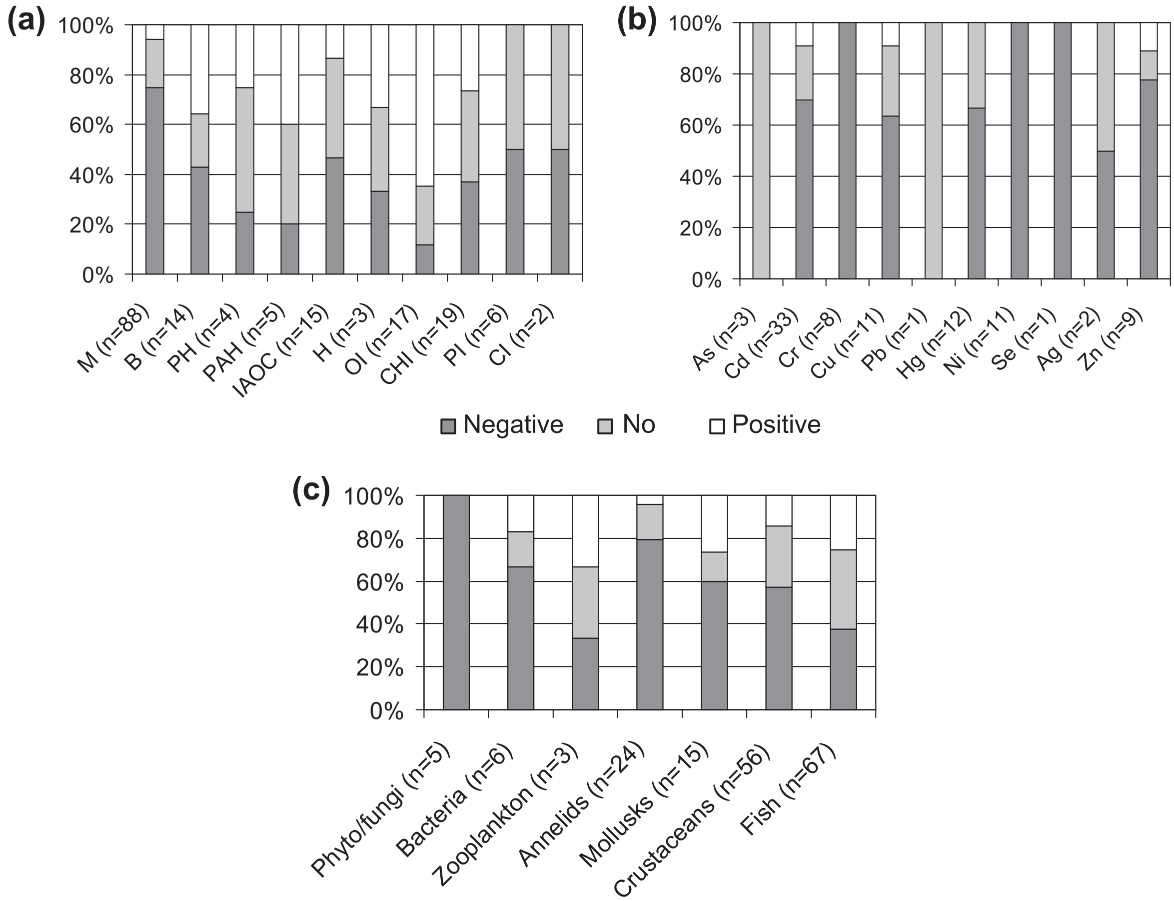

Fig. 3. Frequency of negative (dark grey), no (light grey) and positive (white) correlations between salinity and toxicity of various classes of contaminants (a), metals (b), and all classes of contaminants for the various trophic groups (c), according to Hall and Anderson (1995). Abbreviations: metals (M); biocides (B); petroleum hydrocarbons (PH); policyclic aromatic hydrocarbons (PAH); industrial and agriculture organic chemicals (IAOC); herbicides (H); organophosphate insecticides (OI); Chlorinated hydrocarbon insecticides (CHI); pyrethroid insecticides (PI); and carbamate insecticides (CI).

Table 1
Effects of the temperature on the toxicity and accumulation of metals and organic compounds in aquatic organisms. Abbreviations: *T*, temperature; PCBs, polychlorinated biphenyls; ICo, inorganic Co; CCo, cobalamine Co.

Species	*T* (°C)	Toxic compound		Environment	Reference
		Non-toxic	Toxic (inc. accumulation)		
Paragrapsus gaimardii	5, 19	None	Cd (5 < 19)	Marine	Sullivan (1977)
Corophium volutator	5, 10, 15	None	(As/Zn/Ni) (5 < 10 < 15)	Estuarine	Bryant et al. (1985a),Bryant et al. (1985b)
Macoma balthica	5, 10, 15	None	(As/Zn) (5 < 10 < 15); Ni (all)	Estuarine	Bryant et al. (1985a),Bryant et al. (1985b)
Tubifex costatus	5, 10, 15	None	As (5 < 10 < 15)	Estuarine	Bryant et al. (1985a)
Ophiothrix fragilis	2, 12	Ba/Eu/Ru/Cd/CCo/Zn/Ag	Am (2 > 12)/(ICo/Cs/Mn) (2 < 12)	Marine	Hutchins et al. (1996)
Mytilus edulis	2, 12	(Ag/Am) (12)	(Ag/Am) (2); (Cd/Co/Se/Zn) (2 > 12)	Marine	Baines et al. (2005)
	6, 16, 26	Cu (16, 26)	(Cd/Pb) (6 < 16 < 26); Co(all); Cu(6)	Marine	Mubiana and Blust (2007)
Orconectes immunis	17, 20, 23–24, 27	None	Cu/Zn/Cd/Pb	Marine	Khan et al. (2006)
Palaemonetes pugio	25, 35	None	(Chlorothalonil/Scourge) (25 < 35)	Marine	DeLorenzo et al. (2009)
Sepia officinalis	16, 19	None	(Ag/Cd/Zn) (16 < 19)	Marine	Lacoue-Labarthe et al. (2009)

pared to 2 °C. Selenium uptake from this species was not substantially affected by temperature (Baines, Fisher, & Kinney, 2005; Baines, Fisher, & Kinney, 2006). Mubiana and Blust (2007) evaluated the uptake and accumulation of copper, cobalt, cadmium and lead in *M. edulis* at different temperatures (6–26 °C) and showed a positive relationship between temperature and cadmium and lead accumulation, while cobalt and copper were independent and inversely related to temperature, respectively. In contrast, the elimination process of these toxic metals appeared to be independent of temperature, except for copper. Similar results were also found with cadmium, silver and zinc in the brittle star *Ophiothrix fragilis* at 2 °C and 12 °C (Hutchins et al., 1996). As far as toxicity of toxic metals to marine organisms is concerned, Sullivan (1977) reported that the mortality of the crab *Paragrapsus gaimardii* induced by cadmium was greater at 19 °C compared to 5 °C. Bryant et al. (1985a) reported that arsenic toxicity to three estuarine bottom feeder invertebrates (*Corophium volutator*, *Macoma balthica* and *Tubifex costatus*) increased as temperature and concentration of arsenic increased. Bryant et al. (1985b) evaluated the acute toxicity of nickel and zinc to two estuarine invertebrates (*Corophium volutator* and *Macoma balthica*) at 5, 10 and 15 °C, and found that an increase in temperature caused a decrease in survival for *C. volutator* with both metals, and for *M. balthica* with zinc but not with nickel. A study on the common shore crab *Carcinus maenas* in Stavanger Fjord, Norway, revealed that physiological functions such as heart rate were more vulnerable to copper exposure at seasonal temperature extremes of 5 and 25 °C compared to 15 °C (Camus, Davies, Spicer, & Jones, 2004). Khan et al. (2006) found that the toxicity of copper, zinc, cadmium and lead to the juvenile crayfish *Orconectes immunis* acclimated at 17, 20, 23–24 and 27 °C increased with temperature (by 7–20% between 20 °C and 24 °C and by 14–26% between 20 °C and 27 °C). It is therefore expected that the uptake and toxicity of most toxic metals by marine organisms at different trophic levels will be higher in a warmer ocean.

Increased temperatures normally increase the rate of oxygen consumption in marine organisms; however, copper, zinc, cadmium and lead were found to strongly inhibit the rate of oxygen consumption at all temperatures in the crayfish *O. immunis*, especially for copper, followed by cadmium, zinc and lead (Khan et al., 2006). Temperature can increase the inhibitory effects of toxic metals on respiration of several other marine organisms that utilize the copper-based hemocyanins as respiratory pigments, such as the zebra mussels *Dreissena polymorpha* (Rao & Khan, 2000). At higher temperatures the metabolism of aquatic organisms is increased and oxygen concentration in water is reduced, and therefore, the rate of water inflow into the animal can increase to extract more oxygen, which can increase the entrance of dissolved chemical pollutant/s into the body. Booth and Zeller (2005) modelled the transfer of methyl mercury through the food-web in the marine ecosystem of the Faroe Islands. A 100-year baseline scenario was conducted using only estimated mercury base inflow rate and environmental changes to evaluate changes in fish mortality rates and effects of increased sea temperatures due to climate change on methyl mercury bioaccumulation in all species/groups. The authors estimated that marine organisms at the top of the food chain will face higher accumulation of methyl mercury in the future. In this context, increased temperatures associated with climate change may also increase the sensitivity of marine animals to toxic metals in their environment.

Hypoxia and anoxia induced by climate change may act as covarying stressors with contaminants, compounding the effects of any one of these stressors by them self. Hypoxic episodes (<16% oxygen saturation) can release solid-phase metals like manganese from the sediments increasing concentrations of the bioavailable manganese ion (Mn^{2+}) in bottom waters by 3 orders of magnitude (Schiedek et al., 2007). Manganese is an essential metal, but elevated Mn^{2+} concentrations are toxic to marine organisms, affecting neuromuscular transmission by interacting with the mitochondrial calcium ion (Ca^{2+}) and disturbing the ion balance in muscle membranes (Baden & Eriksson, 2006). Similarly, it was known that mercury biomethylation rapidly occurs in anoxic conditions (Gianguzza, Pelizzetti, & Sammartano, 2000), thus increasing mercury toxicity to marine organisms.

Temperature, salinity and hypoxia can affect the fate and toxicity of toxic metals in marine organisms (bioaccumulation, transport, storage, ligand-interactions, excretion, etc.), and therefore affect seafood safety. The toxicity sites for toxic metals may include mitochondrial and cytosolic enzymes of cellular respiration (Sokolova, Sokolova, & Ponnappa, 2005), antioxidant enzymes (Barata, Lekumberri, Vila-Escale, Prat, & Porte, 2005), respiratory pigments, central nervous system control of respiratory movements (Spicer & Weber, 1991) and ligand-interactions (Amiard, Amiard-Triquet, Barka, Pellerin, & Rainbow, 2006). So far, the synergistic effects of mixtures of toxic metals to marine organisms under various conditions of temperature, salinity and hypoxia are still poorly understood, but it is suspected that the effect might be more-than-additively. For example, Mohan, Gupta, Shetty, and Menon (1986) studied the effect of mixtures of mercury and cadmium on survival, byssus-thread production, oxygen consumption and filtration rate of the intertidal bivalve *Perna viridis*. It was found that the two metals interacted to produce effects that were greater than the sum of the individual effects resulting in mortality in 96 h, and in reducing byssus-thread production, oxygen consumption and filtration rate.

2.2.2. Organic chemicals

Climate change may influence the behaviour and distribution of organic chemical pollutants (OCs) in the ocean, thus causing serious environmental damages and health concerns, particularly at metabolic and physiological levels, both in marine vertebrates and invertebrates (Gordon, 2003). Some anthropogenic chemicals released to the environment can disrupt the endocrine systems of a wide range of wildlife species, particularly the reproductive hormone-receptor systems. Indeed, changes in sperm counts, genital tract malformations, infertility, increased frequency of mammary, prostate and testicular tumours, feminisation of male individuals of diverse vertebrate species and altered reproductive behaviours, have all been reported in specimens contaminated with OCs (Sharpe & Skakkebaek, 1993). In general, OCs are rather stable and toxic, and share a similar structure. Halogenated hydrocarbons chemical stability and lipophilicity and their resistance to degradation results in their persistence in the environment and concentration in food chains in the ocean (Kutz, Wood, & Bottimore, 1991). These substances contain chlorine, iodine, fluorine or astatine, and have low polarity and low water solubility. Aromatic compounds are more reactive and susceptible to chemical and biochemical transformation and include pesticides (chlorinated such as dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethylene (DDE), PAHs, hexacyclohexan, and organometallics like tributyltin). Among organic chemicals, PAHs have become increasingly important because they are potentially mutagenic and carcinogenic in aquatic organisms and humans (Miraglia et al., 2009). The most toxic PAHs are pyrene and fluoranthene, followed by phenanthene and anthracene. PAHs tend to be adsorbed by particles and sink in the sea floor (Miraglia et al., 2009). Since PAHs are quite stable in sediments, bottom dwelling organisms can be exposed to large amounts of PAHs for long periods of time (Neff, 2002). Generally, fish species can excrete PAHs rather fast, therefore having lower body burdens of PAHs compared to shellfish that tend to bioaccumulate such OCs (Neff, 2002). Elevated levels of PAHs are commonly found in coastal and estuarine waters near heavily polluted areas.

The bioavailability and toxicity of OCs in aquatic organisms is likely to increase in response to rising temperature, salinity, hypoxia and ultraviolet (UV) radiation (Heugens, Hendriks, Dekker, van Straalen, & Admiraal, 2001; Schiedek et al., 2007). An underlying mechanism of this interactive toxicity is that temperature alters the toxicokinetics of chemical pollutants in exposed biota (Maruya, Smalling, & Vetter, 2005). Increased temperatures can also alter homeostasis and other key physiological mechanisms, thereby exacerbating the adverse effects of contaminants (Patra, Chapman, Lim, & Gehrke, 2007).

Salinity–contaminant interactions are complex because salinity can influence the chemical itself or it may modulate toxicity and physiological functioning of species (Noyes et al., 2009). OCs are generally less soluble and more bioavailable in saltwater than in freshwater due to strong binding of water molecules and salts inhibiting the dissolution of organic chemicals (Schwarzenbach, Gschwend, & Imboden, 2003). Several studies have evaluated the effect of salinity on the toxicity, uptake and bioaccumulation of

OCs in seafood (see review of Hall & Anderson, 1995). No consistent trend was detected for the toxicity of most OCs with salinity, except with the toxicity of organophosphate insecticides (e.g., parathion, mevinphos, terbufos, trichlorfon), which appeared to increase with increasing salinity (Fig. 3). Generally, euryhaline species are more resistant to toxic conditions at isosmotic salinities (about one third seawater) due to minimization of osmotic stress, and fish are more resistant to toxic chemicals at middle salinities when compared with either lower or higher extremes (Hall & Anderson, 1995; Fig. 3). Consistent trends were not reported from the limited data addressing salinity effects on the toxicity of PAHs to seafood products. A summary of the entire PAH data set shows that toxicity increases at lower salinities in mud crab *Rhithropanopeus harrisii* (Hall & Anderson, 1995) and the estuarine mummichog *Fundulus heteroclitus* (Ramachandran et al., 2006), whereas no trends were observed in grass shrimp (*Palaemonetes pugio*) and tubificid oligochaete worms (*Monopylephorus rubroniveus*) (Weinstein, 2003). Pre-exposing Atlantic salmon smolts to atrazine in freshwater at concentrations greater than 1.0 μg/L resulted in mortality upon a 24-h seawater challenge (Waring & Moore, 2004). DeLorenzo, Wallace, Danese, and Baird (2009) investigated the effects of increased salinity (10 ppt increase) on the toxicity of two common pesticides (the fungicide chlorothalonil and the insecticide Scourge) to the estuarine grass shrimp *Palaemonetes pugio*, and found that the toxicity of chlorothalonil increased with salinity, whereas Scourge toxicity decreased with salinity.

Temperature is one of the main factors able to influence the global distribution and toxicity of OCs (Valole, Codato, & Marcomini, 2007). Monserrat and Bianchini (1995) exposed crabs (*Chasmagnathus granulata*) to methyl parathion and found that there was approximately 10-fold increase in acute lethality with temperature changes from 12 °C to 30 °C. The authors suggested that the higher temperature favours enzymatic activation of this organophosphate insecticide over degradation and excretion. In the estuarine fish, *Fundulus heteroclitus*, warmer temperatures (25 °C) contributed to a rate of elimination of toxaphene congeners that was 2-fold higher than in colder water (15 °C) (Maruya et al., 2005). DeLorenzo et al. (2009) investigated the effects of increased temperature (10 °C increases) on the toxicity of the fungicide chlorothalonil and the insecticide Scourge to the estuarine grass shrimp *Palaemonetes pugio*, and found that the toxicity of both OCs increased with temperature. The studies suggest a general concept for the effect of temperature on toxicity of many contaminants, in which the rates of uptake and excretion generally increases with increasing temperature, whereas the ultimate toxicity of these contaminants depends on whether changes in metabolism result in increased bioactivation or detoxification (Noyes et al., 2009).

Hypoxic conditions also seems to affect detoxification pathways in fish, particularly when exposed to tetrachlorodibenzodioxin (TCDD), which means that its toxicity can increase under hypoxic conditions (Prasch, Andreasen, Peterson, & Heideman, 2004). In the Gulf of Mexico, PAH contamination has been found to reduce oxyregulation capacity in the brown shrimp (*Penaeus aztecus*), suggesting that PAH exposure decreased the ability of *P. aztecus* to survive recent increases in hypoxic events (Zou & Stueben, 2006). This is likely due to disruption of endocrine systems that affect reproduction, similar to the common carp (Wu, Zhou, Randall, Norman, & Lam, 2003) and zebrafish (Shang & Wu, 2004). The inability to cope with hypoxia in combination with exposure to contaminants may result in decreased populations of sensitive species.

Another factor of concern is the increase in UV radiation resulting from the depletion of the ozone layer due to halogen-containing contaminants in the atmosphere, which has been of particular concern in Polar Regions (Schiedek et al., 2007). There is growing evidence that certain PAHs (e.g., anthracene, fluoranthene and pyrene) may pose a greater hazard to aquatic organisms when exposed to UV light due to photo-enhanced toxicity (see Macdonald, Harner, & Fyfe, 2005).

In conclusion, changes in temperature, salinity, oxygen and UV radiation levels may alter the toxicity of certain OCs, and the nature of the such effect will depend on both the organism's life stage and the chemical contaminant.

2.3. Climate and biological contaminants of seafood

Three interlinked factors can affect the susceptibility of organisms to diseases; the organism itself, contaminants, and environment. If any of the three factors are altered, changes in the progression of a disease epidemic can occur. Climate change may impact these factors in various ways, such as by exacerbating the presence of biological contaminants in the marine environment (e.g., toxins produced by harmful algal blooms (HABs)), and increasing populations of pathogenic microorganisms.

Here we provide an overview of some of the most common foodborne illnesses caused by toxin-producing harmful algae and potential impacts of future climate change on these HABs. Climate impacts on infectious diseases acquired from the consumption of seafood contaminated with pathogenic microorganisms are also discussed. These organisms range from viruses to bacteria and protozoan parasites. With the exception of some parasites, most virus and bacteria that cause illnesses from seafood consumption are either introduced during preparation or result from contamination by agricultural or human sources. These forms of seafood contamination are not directly related to climate change. In addition, there is little or no evidence to date to demonstrate that the presence or virulence of fish parasites that impact seafood safety are directly influenced by climate. Therefore, this review will focus only on naturally occurring bacteria and algae (macro- and micro-) in the marine environment that have the capability of being pathogenic to humans.

2.3.1. Harmful algal blooms

The term "harmful algal bloom" is used to describe the proliferation of macroalgae or microalgal cells to densities that cause damage to the environment, threaten the health of humans and aquatic life, and/or alter aquatic food-webs (e.g., Anderson, 1995; Erdner et al., 2008). Although HABs are natural phenomena, their global increase in recent decades has been linked to anthropogenic activities such as cultural eutrophication (or human-caused nutrient enrichment) of water bodies stimulating algal blooms, the transport of harmful algae (HA) species in ships' ballast water, and climate changes (Dale, Edwards, & Reid, 2006; Hallegraeff, 1993; Moore et al., 2008; Tester, 1994; Van Dolah, 2000). Both natural (i.e., the El Niño/Southern Oscillation) and anthropogenic climate changes have been implicated in the observed HAB increase, with warmer than usual conditions generally being linked to increased frequency, duration, and geographic scope of HABs. Projected warming is likely to result in even greater problems due to HABs in the future.

Some HA species produce toxins usually when they bloom. These toxins can accumulate in filter-feeding fish and shellfish and be transferred to organisms higher up in the food chain, such as marine birds and mammals, as well as humans. Toxin-producing HA significantly threaten seafood safety and human health and have significant economic consequences. In addition to contaminating seafood, some HABs can kill fish (e.g., blooms of *Heterosigma akashiwo*); thus, they also impact seafood safety by reducing the availability of fish as a food resource. In the United States alone, the annual economic cost of HABs is conservatively estimated at $82 million (Hoagland & Scatasta, 2006). Approximately half of this cost is due to the public health expenses of HAB-related illnesses, and $3 million is due to monitoring and management efforts to close fisheries during blooms and prevent human intoxications. These costs will increase if HABs continue to increase in the future.

Toxins produced by HA species are generally tasteless and odorless, and heat- and acid-stable. Normal food preparation methods do not prevent intoxication, as opposed to other biological contaminants of seafood such as *Vibrio* spp. that can be killed by cooking. The most common illnesses associated with toxic HABs are ciguatera fish poisoning (CFP), paralytic shellfish poisoning (PSP), amnesic shellfish poisoning (ASP), azaspiracid shellfish poisoning (AZP), diarrhetic (diarrheic) shellfish poisoning (DSP), and neurotoxic shellfish poisoning (NSP). CFP is the most common form of seafood poisoning related to the consumption of finfish, and maybe the common seafood poisoning known (Hokama, 1993; Lange, 1994). The other HAB-related illnesses discussed here result from the consumption of contaminated shellfish. The illnesses, symptoms, toxins, and associated HA species are summarized in Table 2. More detail can be found in a number of reviews of marine toxin seafood poisonings in the literature (e.g., Backer, Fleming, Rowan, & Baden, 2003; Fleming, Backer, & Rowan, 2002). The geographic locations where these illnesses (except AZP) have been reported globally are shown in Van Dolah (2000); however, HAB toxins and their associated illnesses/diseases are increasingly being detected in new regions, which have had no prior reports of illness.

The toxic HA species responsible for the illnesses described above belong to two functional groups of phytoplankton: dinoflagellates and diatoms. Most HA species are dinoflagellates (see Table 2), which possess two whip-like flagella that planktonic forms use for swimming. Under certain oceanographic conditions, this swimming ability gives them a competitive advantage over other non-swimming phytoplankton groups, such as the diatoms. The attributes of these functional groups are important when evaluating their potential responses to future climate change.

Indications of how climate change might impact HABs in the future can be gleaned from examining their responses to climate variations in the past. Large-scale patterns of natural climate variability, such as the El Niño/Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), and North Atlantic Oscillation (NAO), elicit changes in the oceans that are similar to those that are projected to arise from anthropogenic climate change. "Warm" phases of these climate variations range in duration from 6–18 months (i.e., ENSO) to 20–30 years (i.e., PDO). Time series of HA observations with sufficient length to examine the influences of these climate variations are limited, but those that are available provide valuable insight into climate impacts on HABs. As time series of HA monitoring data lengthen, our ability to quantitatively determine climate change impacts on HABs will improve.

Some of the most convincing studies linking climate to HABs outbreaks are for cases of CFP in the tropical Pacific. The HA responsible for CFP are epi-benthic dinoflagellates in the genus *Gambierdiscus* (primarily *G. toxicus*) that live in close association with marine macroalgae in coral reef ecosystems (Hales, Weinstein, & Woodward, 1999). Researchers have found that the abundance of *G. toxicus* and cases of CFP in South Pacific Islands increase during El Niño years (Chateau-Degat et al., 2005; Hales et al., 1999). An El Niño event occurs when the easterly trade winds that blow across the equatorial Pacific weaken and sometimes reverse. This causes water temperatures in the tropical Pacific to become anomalously warm. Approximately 13–17 months after water temperatures are observed to change, *Gambierdiscus* spp. cell density peaks (Chateau-Degat et al., 2005). Human health impacts might be observed three months after the peak in *Gambierdiscus* spp. cell density, indicating that approximately 20 months is required for increased cases of CFP to be observed after climate-induced changes to sea water temperature. The hypothesis that has been put forward to explain this lag is that anomalously warm water temperatures associated with El Niño events causes coral mass bleaching, and the dead coral provides a new surface for the *G. toxicus* to colonize and grow (Chateau-Degat et al., 2005). These results indicate that warming as a result of climate change may increase the incidence of CFP in the South Pacific. Additionally, climate change projections show that the intensity of hurricanes will also increase in the tropical Pacific (Meehl et al., 2007). This may also contribute to increased coral mortality due to physical damage, and create new space that can be colonized by *G. toxicus* (Hales et al., 1999).

Other HABs have also been associated with natural large-scale climate patterns; however, a detailed understanding of how these climate–ocean interactions might directly promote the blooms and/or their toxicity is lacking. For example, in Pacific Northwest region of the United States, increased levels of saxitoxins in the butter clam *Saxidomus giganteus* have been observed during warm/dry climate regimes that are associated with the PDO (Ebbesmeyer et al., 1995). The species believed to be responsible for these toxic events is *Alexandrium catenella*. Recent results indicate that this relationship may arise from warm phases of the PDO increasing the number of days annually that water temperatures are favorable (i.e., >13 °C; Nishitani & Chew, 1984) for *A. catenella* growth and toxin accumulation in shellfish (Moore, Mantua, Hickey, & Trainer, submitted for publication). And in the North Sea, increased abundances of *Prorocentrum* spp. and *Dinophysis* spp. have been associated with warm temperatures and low salinities that occur during warm phases of the NAO (Belgrano, Lindahl, & Hen-

Table 2
Summary of common illnesses caused by toxic marine HABs, symptoms, toxins, and associated HABs species.

Illness	Toxin(s)	Species	Symptoms
Ciguatera fish poisoning (CFP)	Ciguatoxins	*Gambierdiscus* spp.[b]	Nausea, vomiting, diarrhea, numbness of the mouth and extremities, rash, and reversal of temperature sensation. Neurological symptoms may persist for several months
Paralytic shellfish poisoning (PSP)	Saxitoxin and its derivatives	*Alexandrium* spp. *Pyrodinium* spp. *Gymnodinium* spp.	Numbness and tingling of the lips, mouth, face and neck, nausea, and vomiting. Severe cases result in paralysis of the muscles of the chest and abdomen possibly leading to death
Amnesic shellfish poisoning (ASP)	Domoic acid	[a]*Pseudo-nitzschia* spp.	Nausea, vomiting, diarrhea, headache, dizziness, confusion, disorientation, short-term memory deficits, and motor weakness. Severe cases result in seizures, cardiac arrhythmias, coma, and possibly death
Azaspiracid shellfish poisoning (AZP)	Azaspiracid and its derivatives	*Protoperidinium* spp.	Nausea, vomiting, severe diarrhea, and stomach cramps
Diarrhetic shellfish poisoning (DSP)	Okadaic acid and its derivatives	*Dinophysis* spp. *Prorocentrum* spp.[b]	Nausea, vomiting, severe diarrhea, and stomach cramps
Neurotoxic shellfish poisoning (NSP)	Brevetoxins	*Karenia* spp.	Nausea, temperature sensation reversals, muscle weakness, and vertigo

[a] Denotes the only diatom group in the table.
[b] The remaining species are dinoflagellates, with the benthic members.

roth, 1999; Edwards, Johns, Leterme, Svendsen, & Richardson, 2006). In both examples, the climate regime shifts are assumed to have created warmer and more stable water column conditions that are typically thought to favour the growth of dinoflagellates over diatoms (see discussion below on swimming versus non-swimming phytoplankton). A permanent shift to this state as a result of climate change is plausible.

Beyond natural large-scale climate patterns, the potential for climate change (i.e., warmer water temperatures, reduced vertical mixing of waters, changes to the nutrient supply to the upper ocean, and changes to the timing and volume of freshwater runoff entering the coastal ocean) to impact HABs and seafood safety is high. Temperature is an important environmental parameter that regulates many fundamental physiological processes. Eurythermal HA species with wide thermal windows may be favoured by climate change, with warmer temperatures allowing them to expand their geographic ranges into higher latitudes (Pörtner & Farrell, 2008; Tester, 1994). The annual period of time that HABs can develop may also expand as a result of climate change. Warming will allow important temperature thresholds for warm season HA species to be reached earlier in the season and persist for longer periods (Dale et al., 2006; Moore et al., 2008). This may greatly expand optimal growth periods and promote earlier and longer lasting HAB events, as has been shown for some freshwater HA species (Paerl & Huisman, 2008).

The expected reduced rates of vertical mixing of the upper ocean due to climate change will create more stable water column conditions that are typically thought to favour the growth of swimming dinoflagellates over other non-swimming phytoplankton (Solomon et al., 2007). Like all plants, phytoplankton require sunlight and nutrients to grow. In order to obtain adequate sunlight for photosynthesis, phytoplankton need to remain in the illuminated surface waters of the ocean. Under stable water column conditions, the concentration of nutrients in the surface layer can become limiting for growth due to the combination of nutrient uptake by phytoplankton and the reduction of nutrients supplied from below by vertical mixing. When this occurs, dinoflagellates (the phytoplankton group that includes most marine HA) can swim down into the deeper layer to forage for nutrients (Falkowski et al., 2004). Assuming that harmful dinoflagellate species respond to reduced vertical mixing in the upper ocean in a similar way to other dinoflagellates, the frequency of dinoflagellate HABs is likely to increase as a result of climate change.

The responses of marine HABs to other projected impacts of climate change are less certain. For example, upwelling at coastal boundaries has been linked to the development of some HABs (Kudela et al., 2005). Changes to the strength and timing of coastal upwelling winds will change the nutrient supply to the upper ocean and may elicit changes to the patterns of HAB development and occurrence in these systems. The timing and volume of freshwater runoff entering the coastal ocean is also projected to change. For snowmelt dominated rivers, these changes are among the most robust projections of climate change impacts (Barnett, Adam, & Lettenmaier, 2005), but we can only speculate on the responses of HA species to these changes. Increases in freshwater runoff may expose new coastal regions to toxins associated with freshwater HABs, or may enhance water column stability by freshening the surface layer and favour the growth of dinoflagellates over non-swimming phytoplankton over larger areas as buoyant river plumes extend further offshore (GEOHAB, 2006; Sellner & Fonda-Umani, 1999). In contrast, decreases in freshwater runoff might permit marine HABs to occur further upstream (GEOHAB, 2006).

The evidence linking HABs to warmer climate conditions is mounting; however, a "disconnect" between the climate and HAB research communities has thus far prevented the inclusion of future HAB impacts in the IPCC Assessment Reports. This disconnect primarily arises from a mismatch in scales; global scale climate studies typically provide information at 100–300 km resolution, whereas the processes that favour the development of HABs typically occur on local scales of 1–10 km. Elucidating the local processes that give rise to HAB variations, and then developing new methods for downscaling the IPCC climate change scenarios, could be used to build the bridge to HABs impacts studies, including those on seafood safety (Moore et al., 2008).

2.3.2. Pathogens

Many naturally occurring bacteria in the marine environment that have the capability of being pathogenic to humans can be foodborne, usually through shellfish, although many can also be contracted via direct skin contact such as through wounds. The most common and best studied are members of the *Vibrio* spp., a group of bacteria that inhabit ocean and coastal estuarine environments. These bacteria proliferate in a broad range of temperatures, with growth influenced by salinity and other environmental factors. *Vibrio* spp. also have a significant role in the ocean nutrient cycling ecosystem, particularly in the decomposition of chitin (Hunt, Gevers, Vahora, & Polz, 2008). While *V. cholerae* is still the leading cause of Vibrio-associated illnesses worldwide, it is usually transmitted to humans through contaminated water and is not generally considered to be a threat to human health through seafood consumption (Faruque, Albert, & Mekalanos, 1998). Much of what we know about *Vibrio* spp. virulence and adaptive mechanisms in the environment comes from the studies of *V. cholerae* (Faruque et al., 1998). This knowledge is just now being applied to the characterization of two other species of *Vibrio*, *V. vulnificus* and *V. parahaemolyticus*, which cause a significant number of clinical infections, usually through the ingestion of raw or incompletely cooked fish or shellfish (Bonner, Coker, Berryman, & Pollock, 1983; Mead et al., 1999).

V. parahaemolyticus is generally considered to be the most common cause of seafood-borne gastroenteritis in the world (Mead et al., 1999). The disease is usually self-limiting in healthy individuals but severe illness and deaths have occasionally occurred during infections of immunocompromised individuals or after wound infections (Table 3). While first recognized in the 1950s, it was not until 1996 that an increasing number of *V. parahaemolyticus* infections and outbreaks were shown to be caused by strains belonging to a single pandemic clone complex that initially emerged in India and quickly spread to Asia, the North and South American continents (Ansaruzzaman et al., 2005; Chowdhury et al., 2000; DePaola, Kaysner, Bowers, & Cook, 2000; Fuenzalida et al., 2006; Fuenzalida et al., 2007; Gonzalez-Escalona et al., 2005; Martinez-Urtaza et al., 2005; Matsumoto et al., 2000; Okuda et al., 1997; Quilici, Robert-Pillot, Picart, & Fournier, 2005). While *V. parahaemolyticus* infections generally result in relatively mild illness, outbreaks cause considerable loss of consumer confidence in shellfisheries, and therefore have a considerable economic as well as a public health impact.

While not as common as *V. parahaemolyticus*, infections caused by *V. vulnificus* represent a significant percentage of vibrio-related illnesses in the United States (Bonner et al., 1983; Hlady, 1997). As with *V. parahaemolyticus*, *V. vulnificus* infections can be contracted by seafood, usually raw oysters, or through wound infections (Table 3). Overall, infections are relatively rare since susceptibility is generally associated with elevated iron levels in the bloodstream (hemachromatosis or liver cirrhosis) or if an individual is immunocompromised (Strom & Paranjpye, 2000). Even though there are only about 50 cases annually of seafood-related *V. vulnificus* infections, the severity of disease with a $\sim$50% mortality rate makes it the probable leading cause of seafood-associated deaths in the US (Morris, 1988; Morris & Black, 1985).

Table 3
Summary of illnesses caused by *Vibrio spp.* found in shellfish and fish.

Pathogen	Source	Symptoms	Outcome
Vibrio parahaemolyticus	Raw or incompletely cooked fish and shellfish	Diarrhea, vomiting, nausea, abdominal cramps, headache and low fever (Su & Liu, 2007)	Usually self-limiting; death in rare cases where the patient is immunocompromised
Vibrio vulnificus	Raw shellfish	Primary septicemia with sudden onset of fever and chills, vomiting, diarrhea, abdominal pain, and pain in the extremities; secondary cutaneous lesions on the extremities within 24 h, with cellulitis, bullae, and ecchymosis; septicemic shock (Strom & Paranjpye, 2000)	Necrotizing fasciitis, requiring surgical debridement and amputation; primary septicemia; fatal in ~50% of cases

Climate change affecting both global and local environments has the potential of increasing the incidence and spread of several foodborne pathogens, either through the emergence of new pathogens or through the selection of strains of existing pathogens that differ in survival, persistence, habitat range and ability to be transmitted or infect the human host (Gamble, 2008). For example, there is a strong association of sea surface temperature rise with increased occurrence of disease associated with members of the *Vibrio* species (Cook, Bowers, & DePaola, 2002; DePaola et al., 2000). Changes in other environmental factors, such as salinity and pH, may also result in changes in pathogen distribution and virulence through short and long-term adaptive selection of fitter variants of any given species (Elena & Lenski, 2003; Sokurenko, Gomulkiewicz, & Dykhuizen, 2006). Positive selection of a key virulence gene in *V. vulnificus* has recently been demonstrated (Chattopadhyay, Paranjpye, Dykhuizen, Sokurenko, & Strom, 2009), suggesting that adaptive responses to changing environments occur in the *Vibrio* spp.

Genetic studies and genome sequencing efforts have shown that the *Vibrio* genus is extremely plastic at the genome level. Several mechanisms for genetic exchange are active in most species of *Vibrio*, resulting in significant strain diversity and thus in the capability for rapid adaptation and/or expansion into new or changing environmental niches. One major mechanism of genetic exchange involves colonization of chitin-containing substrates, which in turn induces expression of the *pilA*-encoded type IV pilin required for DNA transformation (Meibom et al., 2004). The same gene is also required for DNA transformation of *V. vulnificus* and *V. parahaemolyticus* (Paranjpye & Strom, unpublished results). Recent work has demonstrated that while both *V. vulnificus* and *V. parahaemolyticus* are exceptionally diverse at the genome level, clinical infections may be caused by a relatively few number of strains (González-Escalona et al., 2008; Nilsson, Paranjpye, DePaola, & Strom, 2003; Strom et al., unpublished results; Vickery, Nilsson, Strom, Nordstrom, & DePaola, 2007; Warner & Oliver, 2008). Additionally, it has been demonstrated that populations of *V. cholerae* significantly differ at the genetic level depending on habitat, and that strains populating a specific habitat carried extra genetic information that gave them a selective advantage over strains that had did not have these extra genes (Keymer, Miller, Schoolnik, & Boehm, 2007; Miller, Keymer, Avelar, Boehm, & Schoolnik, 2007). In another study, *V. cholerae* population diversity was directly influenced not only spatially and temporally, but also by pH (Zo et al., 2008). However, environmental factors alone do not always correlate with the presence or absence of these pathogens. For example, total *V. parahaemolyticus* densities were significantly influenced by salinity and turbidity in some Gulf of Mexico oyster harvesting sites but not others (Zimmerman et al., 2007). Interestingly, the number of *V. vulnificus* and *V. parahaemolyticus* infections during the past decade have remained relatively constant or have even increased (Centers for Disease Control, 2009), in spite of education efforts about the risk of raw shellfish consumption, changes in shellfish harvest practices to reduce post-harvest pathogen growth, and the application of post-harvest treatments designed to reduce or eliminate these pathogens from shellfish. In 2004, an unprecedented outbreak of *V. parahaemolyticus* gastroenteritis occurred in Alaska with more than 400 confirmed cases, when cruise ship passengers ate raw oysters harvested from Prince William Sound (McLaughlin et al., 2005). Increased water temperature was considered to be a major factor in the emergence of *V. parahaemolyticus* in Alaska, as the summer of 2004 was exceptionally warm with water temperature remaining above 15 °C over a two month period. However, there was also anecdotal evidence that the strain involved might be more virulent than the majority of strains isolated elsewhere. Genetic characterization demonstrated that most of the Alaskan clinical isolates were phylogenetically distinct from the *V. parahaemolyticus* pandemic complex (González-Escalona et al., 2008). Similarly, strains causing the majority of clinical illness in the Pacific Northwest of the United States during the past 12 years have been shown to be members of a third clonal complex distinct from the pandemic and Alaskan groups (Strom et al., unpublished results). Interestingly, strains highly related to the *V. parahaemolyticus* pandemic complex are prevalent in this same area, yet have not been isolated from clinical infections. Again this suggests that a combination of environmental and genetic factors play a key role in the presence or absence of virulent *Vibrio* spp. Moving forward, other changes in local habitats may contribute to the selection and expansion of more virulent *Vibrio* spp. Most marine *Vibrios* can be found associated with a variety of phytoplankton, zooplankton and copepod species. Changes in densities and environmental niches of these organisms will therefore directly influence the proliferation of *Vibrio* spp. Moreover, ocean acidification will undoubtedly play a role in the selection of strains capable of surviving in a lower pH environment. In addition, ocean acidification significantly alters the ocean's carbonate chemistry, and the resulting impacts on bivalve shell deposition or plankton exoskeleton composition will directly change *Vibrio* spp. habitat, which in turn may force genetic adaptation leading to more virulent strains.

2.4. New and emerging climate-related threats to seafood safety

The effects of climate change on safety of seafood are a relatively new topic. Currently, the issue of vulnerability of seafood to climate change is scarcely considered both at national and at international levels, despite climate change is already affecting the biology and ecology of some organisms, as well as several chemical pathways (Table 4). The issues of seafood security and safety are related because unacceptable standards of food safety that render food unfit for human consumption will also impair seafood security, possibly forcing people to consume seafood that are of lower quality or contaminated, or that have higher (bio)availability of chemical contaminants.

For chemical contaminants, a systematic change in marine hydrographical conditions due to a change towards warmer tem-

Table 4
Effects of climate changes on fate processes for biological and chemical contaminants. Definitions: adherence, is the property of surfaces of different composition to stick together; hydrolysis, chemical reaction in which water reacts with a compound to produce other compounds; photolysis, chemical reaction in which a chemical compound is broken down by photons; biodegradation/transformation, decomposition/breakdown/transformation of a substance through the action of microorganisms (such as bacteria or fungi) or other natural physical processes (such as sunlight); sequestration, formation of a chelate or other stable compound with an ion, atom or molecule so that it is no longer available for reactions; volatilization, vaporization of a dissolved sample; bioconcentration, accumulation of substances in an organism; biomagnification, a cumulative increase in the concentration of a persistent substance in successively higher trophic levels of the food chain.

Fate/process	Impact of climate change
Biological	
Death	Drier summers increase death rate; temperature extremes increase death rate; higher UV radiation levels increase death; flooding and anaerobic conditions increase death
Growth	Increased temperature and wetness increase growth
Lost of active genes	Uncertain
Gene transfer	Uncertain
Adherence	No expected impact
Chemical	
Hydrolysis	Not expected impact
Photolysis	Increases as UV increase rate
Biodegradation/transformation	Higher temperature increase rate; wetter winters increase rate; drier summers decrease rate
Sequestration	Lower for contaminants that sorb to soil organic matter
Volatilization	Increases with increasing temperature
Bioconcentration	Increases with increasing temperature and decrease in salinity
Biomagnification	No expected impact
Dilution	Increases in periods of high rainfall; decreases in prolonged dry periods

peratures, reduced salinity and hypoxia may directly impact seafood safety at several levels (Boxall et al., 2009): (a) increase the inputs of chemicals contaminants to marine systems and the consequent exposure level, particularly due to flood events; (b) change their chemical forms to more toxic ones and the consequent exposure level; (c) increase resuspension processes of sediment-bound chemical contaminants; (d) increase their bioavailability, especially in metals, with contaminants being converted to more bioavailable forms (e.g., increases in temperature enhance the methylation rate of mercury); (e) diminish the species' ability to deal with toxic substances and the different physiological regulation processes involved in the detoxification of hazardous substances; and (f) modification of contaminant transport pathways to marine systems. The significance of a particular pathway or process depends on the underlying properties (e.g., hydrophobicity, solubility, volatility) and form of the contaminant (particulate, particle associated, dissolved, etc.) (Boxall et al., 2009). Several studies indicate that the use of pesticides, other biocides and agrochemicals is likely to increase in the future with climate change due to the expected increase in pest outbreaks in crops, and that more effective pesticides may be required in some instances (Hall, D'Souza & Kirk, 2002; Rosenzweig, Iglesias, Yang, Epstein, & Chivian, 2001). It is anticipated that such practices will result in a higher risk of exposure of seafood and humans to pesticides. Additionally, increased use of chemicals may adversely affect the uptake of toxic trace metals, as shown for cadmium following the application of phosphate fertilizers (Grant, Bailey, Harapiak, & Flore, 2002).

Mercury levels in many parts of the world has reached at the verge of exceeding the threshold level in soil as well as in water bodies and is still increasing because of agricultural runoff and due to sludge produced by industries and population centers (Jan, Murtaza, Ali, Mohd, & Haq, 2009). Recently, researchers in Canada reported for the first time high mercury levels in Canadian Arctic ringed seals and cod that appear to be linked to vanishing sea ice caused by global warming. Seals accumulated more mercury during ice-free seasons that correspond to periods where seals ingest higher amounts of Arctic cod containing high levels of mercury. In this way, longer ice-free seasons resulting from a warming Arctic may result in higher mercury levels in ringed seal populations as well as in their predators (polar bears and humans).

Because climate change is expected to expand the geographic ranges of some harmful microalgae and pathogens, new regions will be exposed to biological contaminants of seafood. Management authorities in these regions may have little or no experience in dealing with these new biological contaminants. Resources will need to be devoted to the provision of adequate training and infrastructure so that monitoring authorities know how to recognize and test for the presence of these harmful organisms in the water and in seafood. For example, as far as algal blooms are concerned, the first shellfish closure in North America due to a toxic bloom of DSP-causing dinoflagellates in the genus *Dinophysis* occurred in March 2008 (WHOI, 2008). Blooms of *Dinophysis* spp. and detections of okadaic acid in shellfish had previously been reported in North America (e.g., Dickey, Fryxell, Granade, & Roelke, 1992; Tango et al., 2002), but had not resulted in a closure. Researchers detected the bloom along the Gulf Coast of Texas while they were monitoring for a different species of harmful microalgae (i.e., *Karenia brevis*) that typically threatens shellfish safety in that region. Because researchers conducting the monitoring program were trained in the identification of harmful microalgae and were aware of the potential impacts of blooms of *Dinophysis* spp., rapid and effective management actions were implemented before contaminated shellfish reached consumers and sickened the public. It is not known if the Texas bloom of *Dinophysis* spp. in 2008 is related to changes in climate, but this example highlights the successful response to a new and emerging threat to seafood safety.

3. Conclusions

In the future, both chemical and microbiological risks may impair seafood safety as a consequence of climate change, in particular algal toxins, organic chemicals residues and toxic metals. Threats to seafood safety due to climate-changes-related alterations in the levels of pathogenic microorganisms in seafood are also relevant. The combination of higher temperatures, lower salinities, hypoxia and ocean acidification will reduce the general fitness of native marine species by changing their physiology, including metabolic rates and enzyme activities. Such changes may have an influence on processes involved in the metabolism of toxic algae, bacteria and chemical substances: higher temperatures result in increased tissue turn-over rates, higher metabolic rates, decrease the species' ability to deal with toxic contaminants and the different physiological regulation processes involved in the detoxification of hazardous substances, increase in the bioavailability of specific contaminants like metals, and increase resuspension processes of sediment-bound chemical contaminants. However, species respond differently to such stresses, e.g. changes in survival patterns of HABs and pathogens, in which some may benefit while others may perish. In this context, outbreaks of chemical and microbial contamination in marine organisms above permissible values are likely to occur more often in the future. Additionally, it is important to recognize that the different changes affecting climate may interact with each other and with other stressors to impact HA and pathogenic bacteria. Predicting the combined impacts of each of these stressors on HA and pathogenic bacteria ecology and toxicology, and on chemical contaminants toxicity, remains a significant challenge. Since seafood is an affordable source of essential nutrients and calories able to prevent cardiovascular and other diseases, public health authorities will face new challenges to guarantee seafood safety and to sustain consumers' confidence in eating seafood.

4. Recommendations and research priorities

We have an opportunity now to limit the negative impacts of climate change in the future. This will require a comprehensive ocean management strategy that incorporates scientific understanding of climate change. This strategy will also require a balance between adaptation to climate change that is unavoidable, and mitigation to reduce the rise in greenhouse gases and resulting impacts.

The seafood industry will be disproportionately affected by climate issues. It is therefore an ethical responsibility that all actors involved in the seafood sector adopt environmentally friendly and fuel-efficient practices to reduce their greenhouse gas emissions, implement sustainable practices to improve the health of our fisheries and aquaculture, avoiding environmental stressors like overfishing, and adopt an appropriate use of veterinary drugs and chemicals in terms of safety, quality, amounts, frequency and timing and withdrawal times. Such practices will enable marine life to adapt to future climate changes. The seafood sector also has the opportunity to mitigate impacts by taking a proactive role in protecting their businesses and way of life, as they have the ability to influence wider business practices and policies. Legislation and principles of good hygiene practices, animal husbandry practices, veterinary practices, aquaculture practices, etc., will have to be adjusted. These changes need to be informed by an enhanced understanding of changes in the occurrence and prevalence of chemical and microbiological hazards in seafood arising from climate change. Food safety is ensured through the implementation of adequate preventive and corrective measures at every step from farm to fork. Integrated monitoring and surveillance of seafood for hazards is critical for the early identification of emerging problems and changing trends in order to protect workers and consumers, and the development and utilization of new seafood preservation technologies to prevent contamination from pathogens post-harvest.

Further research is still required to understand the impact of climate change on the fate of chemical and biological contaminants in the environment, living marine resources and human health. Currently, many scientific studies examine climate sensitivities of species in isolation, and particularly temperature variations. However, other variables also need to be included: (a) ocean acidification; (b) ocean circulation; (c) hypoxia; (d) salinity; (e) UV; and (e) change in cloud cover and sea ice. The next step should examine responses of species populations, communities of multiple interacting species, entire ecosystems, and the combination of multiple stressors (e.g., salinity, temperature, contaminants) to better represent conditions experienced by organisms in nature. Several elements need to be pursued in parallel: improved on-going monitoring of ocean climate and biological trends; laboratory and field process studies; historical and paleoclimate studies of past climate events; and incorporation of the resulting scientific insights into an improved hierarchy of numerical ocean models from species to ecosystems. Good data exchange mechanisms are required at both national and international levels and the data generated should be used in the identification of emerging threats and food contamination trends. Since climate change on local and regional scales is more relevant for people and ecosystems than global trends, improved and better-validated regional ocean climate forecasts remain a major need for future research. Efforts should also be devoted to individualize more precisely the consequences of different climatic scenarios on the related chemical and microbiological issues, in order to better identify adaptation and mitigation measures. Systematic testing is required on the effectiveness and environmental consequences of climate mitigation approaches. Most of these effects should be analysed through an interdisciplinary approach involving expertise from food science, environmental and climatic sciences, economic science, and trade. In this context, long-term strategic national and international investment is required to develop research, monitoring and modelling programs able to enhance climate change knowledge and to develop effective management and mitigation strategies to ensure the safety and sustainability of seafood resources.

Acknowledgments

M.S.S. and S.K.M. acknowledge the support of the Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration (NOAA), and the West Coast Center for Oceans and Human Health (WCCOHH). WCCOHH is funded by the NOAA Oceans and Human Health Initiative, and this work is WCCOHH publication number X.

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