Accepted refereed manuscript of: Groeneveld RA, Bartelings H, Börger T, Bosello F, Buisman E, Delpiazzo E, Eboli F, Fernandes JA, Hamon KG, Hattam C, Loureiro M, Nunes PA, Piwowarczyk J, Schasfoort FE, Simons SL & Walker AN (2018) Economic impacts of marine ecological change: Review and recent contributions of the VECTORS project on European marine waters. *Estuarine, Coastal and Shelf Science*, 201, pp. 152-163.
https://doi.org/10.1016/j.ecss.2016.04.002

Economic impacts of marine ecological change: Review and recent contributions of the VECTORS project on European marine waters

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Abstract
Marine ecological change is likely to have serious potential economic consequences for coastal economies all over the world. This article reviews the current literature on the economic impacts of marine ecological change, as well as a number of recent contributions to this literature carried out under the VECTORS project. We focus on three main types of change, namely invasive alien species; outbreak-forming species, such as jellyfish and toxic algae; and gradual changes in species distribution and productivity. The case studies available in the literature demonstrate that the impacts of invasions and outbreaks on fisheries, aquaculture, and tourism can potentially amount to several tens of millions of dollars each year in some regions. Moreover, stated preference studies suggest a substantial impact on coastal tourism and non-use values that is likely not visible in case studies of specific outbreak events. Climate-driven gradual changes in distribution and productivity of commercial fish stocks will have an impact on fisheries, although these impacts are likely to be overshadowed by much larger changes in prices of seafood and fuel.

1 Introduction

European marine ecosystems are changing under the impact of human activity, and at the same time these changes are having impacts on humans. Introduction of invasive alien species through ballast water, deliberate introductions, and other vectors, are having serious effects on commercial fish stocks and other sectors (see e.g. Xu et al., 2006). Enhanced frequencies of harmful algal blooms (HABs) and jellyfish outbreaks due to overfishing and marine pollution cause considerable damage by killing commercial wild stocks, aquaculture fish and shellfish, deterring or injuring coastal visitors, and making shellfish unfit for consumption (see e.g. Hoagland and Scatasta, 2006; Park et al., 2013). Lastly, climate change, largely driven by anthropogenic greenhouse gas emissions, is having consequences for fisheries as fish stocks move with climatic zones (Doney et al., 2012) and harmful jellyfish and algae are entering new areas (see e.g. Remoundou et al., 2015). The complexity of human drivers and their interlinkages, as well as of the impacts of marine ecological change on humans, calls for an integrated assessment of these impacts and the necessary policy responses.

The VECTORS project aimed to quantify the economic impacts of the wide array of marine ecological changes, and to provide integrated future projections of the social, economic, and ecological changes that might take place in the European marine environment. The variety in drivers, changes, impacts, and disciplinary expertise was addressed by the formulation of two integrated future scenarios for the three EU marine waters considered in VECTORS, namely the Western Mediterranean Sea, the North Sea and the Baltic Sea (Groeneveld et al., this issue). These scenarios were based on the SRES (Special Report on Emissions Scenarios) socio-political storylines used by the IPCC (Intergovernmental Panel on Climate Change) (Nakicenovic et al., 2000) and subsequent studies focused on the marine environment and maritime industries, such as AFMEC (Alternative Future Scenarios for Marine Ecosystems) (Pinnegar et al., 2006) and ELME (European Lifestyles and Marine Ecosystems) (Langmead et al., 2007). These studies formulated four scenarios, from which we adopted two, namely National Enterprise (also indicated as Scenario A2) and Global Community (also indicated as Scenario B1), which we further specified for the Western Mediterranean Sea, the North Sea and the Baltic Sea. Further information on the process of scenario development and the specification of the scenarios can be found in Groeneveld et al. (This issue).

The objective of this article is (1) to review the current literature on the economic impacts of marine ecological change; and (2) to highlight a number of recent contributions to this literature done under VECTORS. We focus on three main types of change, namely (1) marine invasive alien species (IAS); (2) outbreak forming species (OFS); and (3) gradual changes in species distribution and productivity. The geographical distribution of the studies referred to in this review is presented in Figure 1.

The article is organised as follows. Section 2 discusses the economic impacts of marine ecological change, discussing for each of the three main types the current economic literature and the results of VECTORS research. Section 3 discusses economic analyses of policy responses to marine ecological change, including the current literature as well as VECTORS contributions. Section 4 concludes with general observations and suggestions for further research.

2 Economic impacts of marine invasive alien species, outbreak forming species, and gradual changes in species distribution and productivity

2.1 Marine invasive alien species

2.1.1 Estimates in the international peer-reviewed literature

Marine invasive alien species have considerable impact on coastal and marine economic activities, mainly through lost fishing revenues due to predation and competition (Table 1). In 2000 Chinese fisheries suffered economic impacts estimated at US$73.91 million due to exotic species like smooth cordgrass (*Spartina alterniflora*) and several bloom-forming algae (Xu et al., 2006). The Australian spotted jellyfish (*Phyllorhiza punctata*) is estimated to have inflicted up to US$10 million on the shrimp fisheries of the northern Gulf of Mexico in 2000, and is likely to have affected other fisheries as well (Graham et al., 2003). US West coast shellfish fisheries may not yet have felt impacts from the invasive European green crab (*Carcinus maenas*), but its proliferation could lead to annual losses in the order of US$1-2 million (Grosholz et al., 2011). A word of caution, however, is warranted with respect to estimates of direct impacts of invasive alien or outbreak-forming species as an earlier study estimating the impact of *Carcinus maenas* on annual shellfish harvests at US$43 million (Lafferty and Kuris, 1996) may have overestimated *Carcinus maenas*'s economic impacts by an order of magnitude (Hoagland and Jin, 2006).

Occasionally marine invasive alien species present an economic opportunity. In The Netherlands the Chinese mitten crab (*Eriocheir sinensis*) is commercially harvested with a total catch of around 140 tonnes per year at a market price of €12 per kg in 2012 (Bakker and Zaalmink, 2012). In northern Norway the invasive Red King crab has become a highly valuable commercial species (Falk-Petersen and Armstrong, 2013), whereas in Jamaica the invasive Australian Red Claw crayfish appears to be particularly beneficial for poor fishers (Pienkowski et al., 2015). Some exotic shellfish species were introduced for their commercial value (see e.g. Nunes et al., 2004; Troost, 2010): wild specimens of the invasive Pacific Oyster (*Crassostrea gigas*) are harvested in the Dutch Wadden Sea (Van Es et al., 2015).

European seas are reported to contain 879 multicellular invasive alien species (Galil et al., 2015); 176 marine invasive alien species are known to have an economic impact (Vilà et al., 2009). The comb jelly (*Mnempiopsis leidyi*) led to a US$16.7 million reduction in annual anchovy fishery rents in the Black Sea (Knowler, 2005). Frésard and Boncoeur (2006) estimate that controlling the slipper-limpet (*Crepidula fornicata*) in the Bay of St-Brieuc, France will increase the revenues from the local scallop fishery by about € 35.5 million per year.

Table 1: Estimates of economic impacts of marine invasive alien species cited in this review, with their equivalent in 2010 Purchasing Power Parity dollars

Source	Description	Estimate	Estimate in PPP (constant 2010 international $)
Graham et al., 2003	Shrimp harvest lost in 2000 in the Northern Gulf of Mexico due to *Phyllorhiza punctata*	US$ 10 mln	$ 21.28 mln
Knowler, 2005	Lost annual harvests in the Black Sea commercial anchovy fishery due to *Mnempiopsis leidyi*	US$ 16.7 mln	$ 62.02 mln
Frésard and Boncoeur, 2006	Lost annual harvests from scallop fishery due to space competition by *Crepidula fornicata*	€ 35.50 mln	$ 46.60 mln
Xu et al. 2006	Damage to Chinese fisheries due to several exotic cordgrass and bloom-forming algae	US$ 73.91 mln	$ 275.37 mln

	species in 2000		
Grosholz et al., 2011	Lost annual harvest in US West Coast shellfish fishery due to *Carcinus maenas*	US$ 0.62-1.21 mln	$ 0.7-1.4 mln
Bakker and Zaalmink, 2012	Annual *revenues* from harvest of *Eriocheir sinensis* in The Netherlands	€ 1.4 mln	$ 1.65 mln
Börger et al., 2014	Average annual Willingness To Pay of UK citizen for wide spread of invasive species in the Dogger Bank	-£ 25.39	-$ 34.78
This review	Annual medical costs of injuries due to *Crassostrea gigas* in the eastern Scheldt	€ 0.318 mln	$ 0.37 mln
Schasfoort and Van Duinen, 2015	Average annual Willingness To Pay of Wadden Sea tourists for a reduction in *Crassostrea gigas*	Not significant	

2.1.2 VECTORS contributions

VECTORS investigated the impacts of marine invasive alien species in three specific cases: (1) the Dutch part of the Wadden Sea; (2) the Dogger Bank; and (3) biofouling of ship hulls. The Wadden Sea was chosen as a case study because of its combination of high ecological values, a thriving tourist industry, and the abundance of the invasive *Crassostrea gigas*. The Dogger Bank served as a good example of a remote area where construction of offshore wind farms might provide hard substrate for marine invasive alien species. Lastly, biofouling is a relevant issue for shipping in all regional seas considered in VECTORS.

The Pacific Oyster in Dutch coastal areas

A UNESCO world heritage site since 2009, and Europe’s largest marine wetland (Enemark, 2005), the Dutch Wadden Sea is a popular tourism and recreation area. An estimated 13.5 million overnight stays were recorded in 2007 in the area, accounting for €1.4 billion of revenue into the region (Brandt and Wolleson, 2009).

Crassostrea gigas was introduced deliberately in the mid-1960s to the Oosterschelde (Eastern Scheldt), an estuary in the south of the Netherlands (Drinkwaard, 1999; Troost, 2010). In the early 1980s the Pacific Oyster was first observed in the Wadden Sea, after which it spread out all over the Dutch Wadden Sea region (Drinkwaard, 1999). Warm summers contribute to the success of *Crassostrea* (Diederich et al., 2005), so global warming may increase its population. As its population increases, it will take up greater areas of scarce substrate at the expense of the native blue mussel, whose population can accordingly be expected to decline (Troost, 2010).

Crassostrea gigas can potentially impede the Wadden Sea's tourism revenues as its sharp shells cause injuries to tourists engaging in mudflat-walking or water-sports, both of which are popular in the region (Smaal et al., 2006). Although no data are yet available on such injuries in the Wadden Sea, we can approximate the order of magnitude of their medical costs by considering Smaal et al.'s (2006) estimate of about 7000 injuries annually in the Oosterschelde. The standard cost for a consultation with a general practitioner in 2013 was €45.46 (Nederlandse Zorgautoriteit, 2012), which would bring yearly medical cost due to injuries by *Crassostrea gigas* in the Oosterschelde to about €318,220. This should be considered an underestimate of the total economic impact of such injuries, which would also include such issues as transport to the practitioner's office, time lost due to recovery, and the overall discomfort of the injury.

A choice modelling survey (see e.g. Kanninen, 2007) carried out under VECTORS investigated whether tourists are willing to pay to avoid an increase of the Pacific Oyster in the Wadden Sea due to climate change (Schasfoort and Van Duinen, 2015). Experts and practitioners determined the vector of change of *Crassostrea gigas* in the Wadden Sea for (1) the current situation; (2) a situation with limited climate change, corresponding to the Global Community (B1) scenario; and (3) a situation with strong climate change, corresponding to the National Enterprise (A2) scenario. These changes were translated to a visual impact and an increase in the number of mudflat walkers injured, which varied between 1 in 100 (the status quo scenario), 1 in 500, and 1 in 1000 mudflat walkers injured. A daily tourist tax was used as payment vehicle. The choice modelling survey was carried out on the Wadden Sea island Ameland in 2012.

Despite the risk of injury to mudflat walkers due to *Crassostrea gigas*, the coefficient of reducing this risk was statistically insignificant in a conditional logit model and a mixed-logit model, both with and without socio-economic characteristics. This may be due to the relative unfamiliarity of the respondents with *Crassostrea gigas* in comparison with other attributes, such as seals, birds and wind turbines. Furthermore, many respondents did not think an injury due to *Crassostrea gigas* would happen to them. This suggests that despite the medical costs of injuries, the presence of *Crassostrea gigas* does not appear to deter tourists from visiting the Wadden Sea.

Invasive alien species on the Dogger Bank

The Dogger Bank is a shallow water area (less than 20 m in its most shallow regions) of approximately 17,600 km^2 in the North Sea, straddling the borders between the UK, The Netherlands, Germany, and Denmark. The area is a special ecological region with a high fisheries productivity (Kröncke and Knust, 1995; Kröncke, 2011). While plans are being developed to designate the Dogger Bank as a Special Area of Conservation (SAC) under the EU Habitats Directive (EC, 1992), plans have also been submitted to develop an 8,660 km^2

wind farm in the UK sector (Forewind, 2010). The introduction of hard concrete structures, as well as climatic change in general, may facilitate the establishment of marine invasive alien species in the Dogger Bank area (see e.g. Tasker, 2008; Bulleri and Chapman, 2010).

Under VECTORS, Börger et al. (2014) conducted a choice modelling survey among the UK population to estimate the public's appreciation of ecological changes in the Dogger Bank, including the presence of invasive alien species and a household tax as payment vehicle. The valuation scenario used in the choice modelling survey was based on the fisheries management options proposed for the SAC and the offshore wind farm planned for the area. Data collected among 973 respondents were analysed in a conditional logit model and a mixed-logit model. Both analyses suggest a significantly positive willingness to pay (WTP) for increases in general species diversity and the protection of porpoises, seals and seabirds as charismatic species. Based on the results for the mixed-logit model, the annual WTP for wide spread of invasive alien species was estimated at £-25.39 (€-30.29).

The valuation of such remote marine resources, however, bears several types of uncertainty. To decrease cognitive burden on respondents of conveying complex scientific evidence regarding the role of artificial hard structures on the establishment of invasive species (see e.g. Bulleri and Chapman 2010, Dafforn et al. 2015), the valuation scenario merely offered the choice between a wide and restricted spread of invasive species in the North Sea. Any potential uncertainty regarding the consequences of different wind farm configurations was not made explicit because the focus was on examining the welfare effects of *potential* changes rather than the study of how likely these were. Respondents were further given the opportunity to express uncertainty with respect to their stated choices (cf. Brouwer et al.

2010) on a five point scale. Excluding the first and second most uncertain choices (3.4% and 8.3% of all stated choices) from the analysis, however, did not change WTP estimates.

Biofouling of ship hulls

Biofouling creates costs for shipping because hull fouling decreases the speed of ships and increases fuel consumption (already the highest operating cost for the industry) by 40%-80% (Swain et al., 2007). Under VECTORS Fernandes et al. (2016) investigated the costs of hull fouling by indigenous and invasive alien species, as well as possible mitigation methods. For the three European seas studied in VECTORS the study has found indications that non-indigenous species found in ship hulls have a higher impact on fuel consumption (9%-25%) than indigenous species. This is due to factors like a higher average growth (Fernandes et al., 2016) as well as resistance to pollutants (Karatayev et al., 2009) or antifouling coatings (Crooks et al., 2011).

2.2 Outbreak forming species

2.2.1 Estimates in the international peer-reviewed literature

Indigenous outbreak-forming species, such as jellyfish and harmful algae, have a variety of economic impacts, such as losses in fishing and tourism revenues, health care costs, and recreationists' appreciation of beach visits. The main impacts of jellyfish blooms include (1) predation of commercial fish stocks and clogging of fishing nets; (2) killing of penned fish in aquaculture; (3) impacts on tourism; and (4) blockage of sea water intake by power stations (Purcell et al., 2007). Most economic valuations of jellyfish impacts regard effects on fisheries (Table 2). Mass occurrence of the giant jellyfish (*Nemopilema nomurai*) in Japan in 2003 caused US$ 20 million worth of damages to fisheries in Aomori prefecture alone (Kawahara et al., 2006). Impacts on Korean fisheries are estimated at annually US$ 68.2

million to US$ 204.6 million, which includes reductions in catch as well as reduced value of the remaining catch (Kim et al., 2012). Jellyfish are frequently caught as bycatch in the Peruvian anchovy fishery, and quality control managers deduct the amount of jellyfish caught from total catch. A 2013 survey (Quiñones et al., 2013) among quality control managers in the Peruvian anchovy fishery estimated the economic losses due to such deductions at about €200,000 in the 35 days of the survey. Finally, Palmieri et al. (2014) estimate the economic losses of the trawling fleet in the northern Adriatic Sea due to reduced harvests at €8.2 million per year, whereas additional fuel costs of €460,000 were made as fishers needed to fish further from port to avoid jellyfish.

Recent years have seen a blooming of non-market valuations of jellyfish impacts, focusing on beach visitors and local residents. Kontogianni and Emmanouilides (2014) estimates in a choice modelling survey that residents and tourists along the coast of the Gulf of Lion are willing to pay between €59.07 and €68.79 per household (either as a one-off increase in next year's water bill or a one-off fee paid to hotels) to reduce the frequency of jellyfish outbreaks from nine in ten years to only one in ten years. Ghermandi et al. (2015) combined the contingent behaviour and contingent valuation methods to investigate the economic impacts of jellyfish on beach tourism near Tel Aviv. The study estimates that a jellyfish bloom reduces beach visits by 3%-10%, which corresponds to a monetary loss between €1.8 million and €6.2 million. In a choice modelling survey among residents of the Bay of Santander area, Remoundou et al. (2015) investigate respondents' WTP for reducing the frequency of beach closures due to the occurrence of *Physalia physalis*[1], which is currently estimated at 15 days per year. With an annual payment for the coming five years as bid vehicle, the authors

[1] Although *Physalia physalis* is taxonomically not a true jellyfish but a colony of siphonophores we will treat it as a jellyfish in this review.

estimate that respondents are willing to pay €25.23 and €30.33 for a reduction to 10 days per year and 5 days per year, respectively.

Toxic algae are a health risk for shellfish consumers and tourists, and have a direct impact on wild and cultured fish and other commercial marine animals (Table 3). Landsberg (2002) provides a global overview of HAB events and their economic impacts on fisheries and aquaculture. The most dramatic impacts were recorded in relation to an outbreak of *Noctiluca scintillans* in the Pei Hai Sea with impacts of US$ 100 million to shrimp mariculture (Chen and Gu, 1993), and an outbreak of *Cochlodinium polykrikoides* in Korea that caused losses to fisheries estimated at US$95.5 million (Kim et al., 1999). A later study estimated the latter outbreak's impact on aquaculture at US$60 million (Park et al., 2013). Hoagland and Scatasta (2006) estimate the average annual economic effects of HABs at 2005 US$813 million in the European Union and 2005 US$82 million in the United States; these effects largely regard losses of revenues in the tourism and shellfish industries and the medical costs of shellfish poisoning. A later study by Hoagland et al. (2014) suggests that the annual costs of respiratory and digestive illness along the Gulf Coast of Florida caused by blooms of *Karenia Brevis* amount to US$60,000 to US$700,000. Larkin and Adams (2007) studies the economic impacts of HABs on local businesses in two zip codes in Florida and estimates that on average a red tide event depresses local restaurant and lodging revenues by about US$6.5 million per month. Jin et al. (2008) describe a bloom event in New England that led to widespread closures of shellfish fisheries, with an economic impact ranging from at least US$2.4 million in Maine to up to US$18 million in Massachusetts. Dyson and Huppert (2010) studies the direct impact of closures of the recreational razor clam fishery in Washington State, as well as its wider impacts throughout the local economy. The authors estimate that closing an average harvest opening (4-5 days) costs about 2008 US$4 million, whereas closing the harvest for an entire year costs about 2008 US$20.4 million.

Stated preference studies on HABs have estimated respondents' WTP at €9.73 at once for a programme that keeps Bulgarian beaches free of algal blooms (Taylor and Longo, 2010); €76 per year for prevention and control measures against HABs in the province of North Holland, The Netherlands (Nunes and van den Bergh, 2004); and up to €666 per year in Finland to reduce the biomass of cyanobacteria by up to 35% (Kosenius, 2010). Even taking into account the difference in price level between Bulgaria and Finland (a conversion factor of 2.4 in 2010-2014: World Bank, 2015), this suggests a wide variety in respondents' WTP for reducing the frequency of HABs. The effects on the local economy in coastal recreational regions may also be substantial: Morgan et al. (2010) estimate in a choice modelling survey that about 70% of respondents would change their beach going plans in the event of a red tide.

2.2.2 VECTORS contributions

Within VECTORS two choice modelling surveys have investigated the impact of HABs on recreation values. The first estimates the impacts of jellyfish on the Spanish coast of Catalonia, whereas the second focuses on the impacts of filamentous algae in the Gulf of Gdańsk.

Jellyfish along the Catalan coast

So far economic valuations of losses in recreational value due to jellyfish blooms have been rare. Under VECTORS, Nunes et al. (2015) addressed this gap for the coast of Catalonia, Spain, by estimating how much beach visitors are willing to spend, in terms of reported extra travel time, for lowering the risk of jellyfish outbreaks. The Catalan coast constitutes a world-leading coastal tourist destination with 263.7 million registered recreational beach visits in 2012, so its economy may be severely impacted by jellyfish outbreaks. Nunes et al. (2015) carried out a choice modelling survey in which 644 respondents were presented with a number of choice sets between a beach destination with a "low" risk of jellyfish blooms (two days per week or less), a beach destination with a "high" risk of jellyfish blooms (five days per week or more), or not going to the beach at all. Other attributes of the beaches included water transparency, services, and additional travel time (the payment vehicle). Data were analysed with a conditional logit model and a random parameter logit model for the main effects and interactions between jellyfish risk and survey characteristics (whether the survey was taken on Blanes, which is a particularly low-risk beach; whether the respondent ever has been stung by jellyfish; whether the respondent is a local resident). The results suggest that respondents were willing to spend on average an additional 23.8% of their travel time to visit a beach where jellyfish outbreaks occur in two days per week or less. The authors estimate the annual economic gains associated with reduction of jellyfish outbreaks on the Catalan coast of around €422.57 million, or about 11.95% of tourism expenditures in 2012.

Filamentous algae in the Gulf of Gdansk

Outbreaks of filamentous algae in the Gulf of Gdańsk in Northern Poland, which mainly includes brown algae (*Ectocarpus*, *Pilayella*), are driven by increased eutrophication, high temperature, and calm windless weather. In such conditions, filamentous algae grow fast on hard substrata and create algal mats, which are detached and float towards the shore. They create problems to seagrass (*Zostera marina*) meadows as they cover the seagrass leaves, reduce light and mechanically disturb plants. These mats are also a problem for recreational beaches as they easily decay and form local anoxic conditions. When *Zostera* meadows are dense and healthy, they can easily compete with filamentous algae for nutrients, and as a result they inhibit the development of outbreak species. When seagrass is weak, filamentous algae benefit as they are faster in nutrient acquisition (Węsławski et al., 2013).

So far, few studies have attempted to estimate the monetary value of seagrass restoration, nor the economic impact of filamentous algae. Within VECTORS, Börger and Piwowarczyk (Forthcoming) conducted a choice modelling survey which aimed at assessing the value of expanding underwater seagrass meadows. Direct interviews with households in the region adjacent to the Gulf (N=500) were employed to elicit preferences regarding a seagrass restoration programme. The payment vehicle regarded a waste water treatment fee, and the survey data were analysed with conditional logit and mixed-logit models. Besides values for water purification and the opportunity for recreation and tourism provided by seagrass, the study found significant WTP of respondents for a reduction of filamentous algae (*Ectocarpus* and *Pilayella*). While annual WTP for a reduction of filamentous algae from 30,000 to 10,000 tons per year is €14.48, WTP for a larger reduction down to 1,000 tons is not significantly larger (€15.63). A possible explanation is the small difference in impact as explained in the survey: a reduction from 30,000 tons to 10,000 tons eliminates the immediate nuisance as no more algae are present on and close to the shore, whereas a further reduction to 1,000 tons only affects regions further away from the shore.

Table 2: Estimates of economic impacts of jellyfish outbreaks cited in this review, with their equivalent in 2010 Purchasing Power Parity dollars

Source	Description	Estimate	Estimate in PPP (constant 2010 international $)
Kawahara et al.,	Damages to fisheries in Aomori prefecture,	US$ 20 mln	$ 19.06 mln

2006	Japan, in 2003 due to *Nemopilema nomurai*		
Kim et al., 2012	Losses in fisheries revenues in 2006-2010 in Korea due to jellyfish species such as *Aurelia aurita* and *Nemopilema nomurai*	US$ 68.2-204.6 mln	$ 95.38-286.13 mln
Quiñones et al., 2013	Lost revenues in Peruvian anchovy fishery due to several jellyfish species in 35 days of survey in port of Ilo	US$ 0.2 mln	$ 0.43 mln
Kontogianni and Emmanouilides, 2014	Average one-off Willingness To Pay per respondent for reducing frequency of jellyfish outbreaks in the Gulf of Lion from nine to one in ten years	€ 59.07-68.79	$ 69.69-81.16
Palmieri et al. 2014	Reduction in annual fish catch plus avoidance costs in the northern Adriatic trawling fleet due to jellyfish such as *P. Noctiluca*	€ 8.66 mln	$ 11.09 mln
Ghermandi et al., 2015	Loss in beach recreational consumer surplus in Tel Aviv due to *Rhopilema nomadica*	ILS 8.9-31.1 mln	$ 2.06-7.19 mln
Remoundou et al., 2015	Annual WTP per respondent for reduction of Santander beach closures due to *Physalia physalis* from 15 days per year to 10 and 5, respectively	€ 25.23; € 30.33	$ 39.28; $ 47.22
Nunes et al., 2015	Estimated total annual Willingness To Pay of Catalan coast visitors for reduction in jellyfish prevalence	€ 422.57 mln	$ 587.25 mln

Table 3: Estimates of economic impacts of harmful algae cited in this review, with their equivalent in 2010 Purchasing Power Parity dollars

Source	Description	Estimate	Estimate in PPP (constant 2010

			international $)
Chen and Gu, 1993	Losses in Pei Hai Sea shrimp mariculture in 1989-1990 due to *Noctiluca scintillans*	US$ 100 mln	$ 463.73 mln
Kim et al., 1999	Losses in Korean fisheries in 1995 due to *Cochlodinium polykrikoides*	US$ 95.5 mln	$ 171.16 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on public health in the European Union	US$ 11 mln	$ 12.40 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on commercial fisheries in the European Union	US$ 147 mln	$ 165.74 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on recreation and tourism in the European Union	US$ 637 mln	$ 718.19 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on public health in the United States	US$ 37 mln	$ 40.72 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on commercial fisheries in the United States	US$ 38 mln	$ 41.82 mln
Hoagland and Scatasta, 2006	Annual economic impacts of HABs on recreation and tourism in the United States	US$ 4 mln	$ 4.40 mln
Larkin and Adams, 2007	Loss in local restaurant and lodging revenues in Ft. Walton Beach and Destin, Florida, in the event of a *Karenia brevis* bloom	US$ 6.5 mln	$ 8.73 mln
Jin et al., 2008	Lost shellfish fishery revenues in coastal Maine due to a 2005 *Alexandrium fundyense* outbreak	US$ 2.4 mln	$ 2.64 mln
Jin et al., 2008	Lost shellfish fishery revenues in coastal Massachussetts due to a 2005 *Alexandrium fundyense* outbreak	US$ 18 mln	$ 19.81 mln
Dyson and Huppert, 2010	Regional economic impact of beach closures of 4-5 days due to *Pseudo-nitzschia* and *Alexandrium* outbreaks	US$ 4 mln	$ 4.08 mln

Dyson and Huppert, 2010	Regional economic impact of beach closures of 1 year due to *Pseudo-nitzschia* and *Alexandrium* outbreaks	US$ 20.4 mln	$ 20.81 mln
Kosenius, 2010	Average annual Willingness To Pay per household for a reduction of cyanobacteria biomass by 15-35% in the Gulf of Finland	€ 210-666	$ 249.69-791.86
Taylor and Longo, 2010	Average one-off Willingness To Pay per respondent for elimination of HABs in Varna Bay, Bulgaria	BGN 18.97	$ 38.35
Park et al., 2013	Losses in Korean aquaculture due to a 1995 *Cochlodinium polykrikoides* bloom	US$ 60 mln	$ 108 mln
Hoagland et al., 2014	Annual costs of illness due to *Karenia brevis* outbreaks in the Florida Gulf Coast	US$ 60,000-700,000	$ 56,901-663,846
Börger and Piwowarczyk, Forthcoming	Average annual Willingness To Pay for reduction of filamentous algae in the Gulf of Gdansk from 30,000 to 10,000 tons per year	€14.48	$ 30.93

2.3 Gradual changes in species distribution and productivity

2.3.1 Estimates in the international peer-reviewed literature

Global change in the climatic and biophysical environment will likely affect marine ecosystems in a variety of ways, including rising temperatures, enhanced stratification, dropping oxygen levels, and declining pH (Doney et al., 2012). These changes will affect commercial fish stocks, which may shift towards the poles or to deeper depths. Moreover, rising temperatures and eutrophication can potentially worsen the aforementioned issues of marine invasive alien species, HABs, and jellyfish blooms (Stachowicz et al., 2002; Richardson et al., 2009; O’Neil et al., 2012).

Sea level rise is prominent in the literature on the economic effects of climate change on coastal zones, and it is projected to cause damages in the order of magnitude of billions of US dollars (Bosello et al., 2007; Bosello et al., 2012; Hinkel et al., 2014). Available studies of the economic impact of climate change on fisheries are patchy (Sumaila et al., 2011). Garza-Gil et al. (2011) estimates that annual profits in the European sardine fishery may drop from about € 38 million in 2010 to about € 30 million 2030 (in the price level of 2010), as sardines are sensitive to rising temperatures. Assuming the recommended social discount rate in EU cost-benefit analyses (EC, 2014) of 3%, the losses projected in Garza-Gil et al. (2011) between 2010 and 2030 amount to a present value of about € 64 million. Although ocean acidification is likely to have severe economic impact on fisheries and tourism (Rodrigues et al., 2013; Voss et al., 2015), the information necessary to economically assess these impacts remains limited (Hilmi et al., 2013).

2.3.2 VECTORS contributions

Economic research within VECTORS investigating gradual changes in species distribution and productivity focused on three main issues: (1) impacts on fisheries of spatial shifts of commercial fish populations; (2) impacts on the presence of charismatic species; and (3) economy-wide impacts of climate change through impacts on fisheries and tourism.

Impacts on fisheries of spatial shifts of commercial fish populations

Impacts on fisheries were analysed with the bioeconomic models FISHRENT (Salz et al., 2011; Simons et al., 2014) for the North Sea saithe fishery, SIMFISH (Bartelings et al., 2015; Bartelings and Hamon, 2015) for the North Sea flatfish and shrimp fishery, and MEFISTO (Maynou, 2014) for the Mediterranean fisheries. The analyses focused on the ecological, political and economic changes between 2010 and 2050 according to the scenarios specified in Groeneveld et al. (This issue).

Saithe is of major economic importance for North Sea fisheries, with landing values in the German fleet of 11.1 million Euros in 2013 (STECF, 2014). The North Sea saithe fishery also represents a problematic case, however, due its substantial catch of cod, which Total Allowable Catch (TAC) in the last two years was about a third of that of saithe in the North Sea and Skagerrak (ICES, 2013, 2014), indicating that the cod quota may be exhausted faster than the saithe quota. Simons et al. (2014) simulated the degree of spatial overlap of saithe and cod under a northward shift of cod (e.g. Engelhard et al., 2014). The study found that decrease of profits is less in Global Community (B1) (about €50,000, i.e. 3% of current profits) than in National Enterprise (A2) (about €133,000, i.e. 8% of current profits) because of higher stock spawning biomass (SSB) of saithe, lower fishing effort and lower fuel prices in B1. The higher SSB of saithe in B1 is partly caused by a seasonal closure of the spawning ground of saithe assumed in the B1 scenario (see Groeneveld et al., this issue). Average fishing effort between 2007 and 2050 is 13% higher for A2 and 12% lower for Global Community (B1) than in the present situation. This is caused by the fact that on average SSB of saithe is 19% lower for A2 and 6% higher for B1 than in the present situation.

Compared with a base scenario with no changes in policy, world fuel and food prices, or biological parameters, the SIMFISH results for the North Sea flatfish fishery show an increase in profits in the A2 scenario and a decline in the B1 scenario (Bartelings and Hamon, 2015). Under a 3% discount rate (EC, 2014), the present value of profits projected between 2010 and 2050 are about € 538,000 for the base scenario, € 581,000 for the A2 scenario, and € 459,000 for the B1 scenario, respectively. World fuel and fish prices have the biggest impact of all factors included in the model. Keeping all other factors as in the base scenario, the projected displacement of sole and plaice to northern areas of the North Sea reduces projected profits by between 1% (A2) and 2% (B1); closing parts of the North Sea to fishing has a somewhat bigger impact (-3% and -7% for A2 and B1, respectively). In the A2 scenario the increase in fish prices more than compensates for these losses, as it would increase profits by 16% if all other factors were kept equal; in the B1 scenario rising fuel prices lead to a decrease in profits by 3%. It should be noted, however, that the joint impact of these three types of changes differs from the sum of impacts of the three isolated changes.

The main fishing fleets in the Western Mediterranean are bottom trawlers and purse seiners. Projections with the MEFISTO model of annual profits in 2050 suggest that profits are much higher in the B1 scenario than in the A2 scenario, partly because of stricter fishing policy and partly because of lower fuel prices. For purse seine fleets annual profits are projected at € 16 million (A2) and € 49 million (B1), compared to the base scenario were profits are estimated at € 6 million. For the trawl fleet the projections suggest annual losses of € 46 million (A2) and annual profits of € 329 million (B1) compared to current annual profits estimated at about € 200,000. This large difference in projected profits is likely due to the high sensitivity of trawl fleets to fuel prices.

Effects on charismatic species

Two choice modelling surveys carried out under VECTORS estimated the non-use values of charismatic species on the Wadden Sea and the UK part of the Dogger Bank. The Wadden Sea study (Schasfoort and Van Duinen, 2015) considered seals and birds in a choice modelling survey to assess whether tourists are willing to pay to avoid a decrease in number and diversity of these species due to climate change. Although there is no agreement yet about the impact of climate change on birds and seals in the Wadden Sea, the study distinguished, together with experts, two possible effects of climate change on these species. First, mud flats in the Wadden Sea may not keep pace with sea level rise (Wang et al., 2012), which may impact hibernating places for birds and nursery places for seals. Second, birds may face a decrease in food availability (van Roomen et al., 2012; Clausen et al., 2013). This has been translated in three different attribute levels corresponding with the current situation, a low climate change scenario and a high climate scenario. Tourists were asked to state their WTP to avoid a high climate change scenario.

Results from a mixed-logit model indicate that tourists are willing to pay the most to obtain a stable bird population instead of a decreasing population (€ 7.72 per tourist per day), followed by a limiting the decrease in number and diversity of birds to a small decrease (€ 5.72 per tourist per day). In addition, tourists were willing to pay € 4.72 per day for a stable seal population, whereas the WTP was € 3.24 per tourist per day to obtain an increase in abundance instead of a decrease. What is remarkable is the preference of tourists for the stabilisation of the seal population instead of growth, suggesting that tourists prefer a balanced ecosystem to seeing a seal more frequently.

The UK choice modelling survey by Börger et al. (2014) found that the value of charismatic species by far exceeds the value of general species diversity. Members of the general public were willing to pay £24.02 (€28.66) annually to protect porpoises, seals and seabirds on 25% of the UK's Dogger Bank area. This contrasts with a WTP of only £4.19 (€5.00) for a 10% increase in general species diversity. The WTP for the protection of charismatic species on 50% of the UK's section of the Dogger Bank, however, was £30.32 (€36.17).

The WTP for conservation of charismatic species groups estimated in these two choice experiments appears to be near the lower end of the range of values for threatened or endangered species found elsewhere in the literature (Richardson and Loomis, 2009). Two caveats are warranted, however, when comparing these studies. First, the *annual* WTP found by Börger et al. (2014) for conservation of charismatic species is only one order of magnitude higher than the *daily* WTP found by Schasfoort and Van Duinen (2015) for different, albeit still charismatic species groups. This could be the result of respondents' weak scope sensitivity or genuine diminishing returns to scale of charismatic species conservation (see e.g. Lew and Wallmo, 2011), or of other differences between the two cases. Similar differences between monthly and annual payments are also reported by Richardson and Loomis (2009). Second, the values reported in most publications, notably those cited by Richardson and Loomis (2009), regard single species, whereas Börger et al. (2014) and Schasfoort and Van Duinen (2015) value conservation of species groups.

Economy-wide impacts

Computable General Equilibrium (CGE) models are increasingly used in the economic assessment of climate change impacts (see e.g. Darwin and Tol, 2001; Bosello et al., 2012; Eboli et al., 2010; Ciscar et al., 2011). These multi-sector, multi-country models describe how economic consequences of climate change spread internationally and intersectorally, and estimate the final GDP and welfare effects. Within VECTORS the recursive dynamic CGE model ICES is applied to assess the medium-term (2030) economic effect of future changes in the EU marine ecosystem in the Western Mediterranean, the North Sea and the Baltic Sea. The ICES model was calibrated to replicate for the period 2010-2030 the population and GDP growth rates of the SRES A2 and B1 scenarios (Nakicenovic et al., 2000), on which the VECTORS scenarios are based (Groeneveld et al., this issue). Focusing on the fishing and the tourism sectors, important variables for the analysis are projected fuel and fish prices, as well as changes in stock abundance. Therefore, projections for economic variables in scenarios A2 and B1 were "perturbed" in the ICES model for a decrease in fish stock abundance due to overfishing and natural drivers related to climate change, such as invasive species.

The results suggest that Gross Domestic Product of EU coastal states is negatively affected with larger losses associated with decreases in tourism demand. This is explained by the much higher contribution of tourism to value added than fisheries. Impacts are generally more negative in the A2 scenario than in the B1 scenario. The largest absolute GDP losses in 2030 related to fishing activity compared to the baseline are experienced by France (A2: $ 4.1 bln; B1: $ 2.2 bln). It should be noted however, that the fishing sector contributes a small section of national GDP in all countries concerned so that compared to the overall size of the economy these losses are less than a half percent for all countries considered. The biggest absolute losses related to tourism are experienced by Germany (A2: $ 37.7 bln; B2: $ 30.9 bln); Italy suffers the biggest losses compared to the overall size of its economy (about 1% in both scenarios).

CGE models are helpful to highlight and quantify the role of indirect economic effects and the overall order of magnitude and direction of the phenomena analysed, but the results should not be considered as exact quantifications. They assume perfectly competitive markets, perfectly rational agents and exogenous technological change, and thus represent ideal situations far from real-life economic systems. Furthermore their evaluations are based upon GDP which records just what occurs in measurable market transactions and thereby excludes such economic activity as volunteer work, housework, and illegal transactions. Nevertheless, these results indicate that changes in marine ecosystems can have wide macroeconomic effects, and, although these may appear to be small in terms of national GDP, they are far from negligible at the sectoral level.

Table 4: Estimates of impacts of gradual changes in species distribution and productivity cited in this review, with their equivalent in 2010 Purchasing Power Parity dollars

Source	Description	Estimate	Estimate in PPP (constant 2010 international $)
Garza-Gil et al., 2011	Present value (discount rate 3%) of profits losses in the European sardine fishery projected over 2010-2030 due to rising temperatures	€ 64 mln	$ 76 mln
Börger et al., 2014	Annual willingness to pay to protect porpoises, seals, and seabirds on 25% and 50% of the UK part of the Dogger Bank	£24.02; £ 30.32	$ 32.90; $ 36.17
Börger et al., 2014	Annual willingness to pay for 10% increase in species diversity in the UK part of the Dogger Bank	£ 4.19	$ 5.00
Buisman et al., 2014	Change in profits in the Western Mediterranean trawl fisheries between 2015 and 2050 under scenarios A2 and B1	A2: -€ 46 mln[a] B1: € 329 mln[a]	A2: -$ 60 mln B1: $ 426 mln
Buisman et al., 2014	Effects on the Western Mediterranean purse seine fisheries under scenarios A2 and B1	A2: € 10 mln[a] B1: € 43 mln[a]	A2: $ 13 mln B1: $ 56 mln
Simons et al., 2014	Decrease in annual profits of North Sea saithe fishery under scenarios A2 and B1	A2: € 133,000[a] B1: € 50,000[a]	A2: $ 153,267 B1: : $ 57,619
Bartelings and Hamon, 2015	Present value (discount rate 3%) of change in profits North Sea demersal fisheries under scenarios A2 and B1	A2: € 43,000[a] B1: -€ 79,000[a]	A2: $ 51,293 B1: -$ 94,236
Schasfoort and Van Duinen, 2015	Tourists' daily willingness to pay for (a) limited decrease in bird populations; (b)	(a) € 5.72 (b) € 7.72	(a) $ 12.22 (b) $ 16.49

	stabilisation of bird populations; (c) stabilisation of seal populations; (d) increase in seal populations	(c) € 4.72 (d) € 3.24	(c) $ 10.08 (d) $ 6.92
ICES model	Loss in GDP in the EU between 2015 and 2030 due to ecological change under scenarios A2 and B1	A2: $ 36.4 bln B1: $ 32.2 bln	A2: $ 36.4 bln B1: $ 32.2 bln

[a] Joint impact of ecological, economic, and policy changes

3 Management responses

3.1 Prevention of marine invasive alien species

Introductions can to some extent be prevented through ballast water treatment and controlling biofouling (Reise et al., 1998; Minchin and Gollasch, 2003; Olenin et al., 2010). Fernandes et al. (2016) estimated that the costs of regular maintenance and cleaning to avoid hull fouling can amount up to 5%-10% of annual costs (operational and capital amortization costs) for smaller vessels and 1%-3% for larger vessels; these cost estimates, however, do not include the disturbance of fleet operations by maintenance and cleaning. The costs of ballast water treatment systems (BWTS) are estimated to be on average between 1.4% and 2.9% of annual operating costs depending on the type of ship (Fernandes et al., 2016). The proportion of costs of BWTS of total shipping costs can be higher for some types of smaller ships (3.9%-9.9%) than for bigger ships (Smith, 2013; Fernandes et al., 2016).

3.2 Mitigation and control of invasive and outbreak-forming species

Eradication of invasive species is usually exceedingly difficult, but rational policies can entail reasonable control to limit their impacts. In their bioeconomic analysis of the management of the invasive slipper-limpet (*Crepidula fornicata*) in the Bay of St-Brieuc, France, Frésard and Boncoeur (2006) estimates that the aforementioned increase in annual scallop harvests of € 35.5 million due to control of the species (also see Table 1) can be attained at an annual control cost between € 4.3 million and € 6.9 million, depending on the assumed harvest function.

Richardson et al. (2009) describes a wide range of short-term control measures of jellyfish, including destroying jellyfish with cutting nets, removing polyp beds by cleaning artificial hard structures, using biocontrol agents, and preventing introduction of invasive jellyfish through hull-cleaning and ballast water treatment. The article also indicates, however, that many questions remain regarding the effectiveness and possible side-effects of these measures, which possibly explains the paucity of economic analyses of jellyfish control measures in the scientific literature. In the long term reducing eutrophication, reducing overfishing, and minimising global warming will be needed to prevent jellyfish blooms from forming.

The VECTORS choice modelling survey carried out in Catalonia (Nunes et al., 2015) underlined the urgency to provide daily information with social media applications or other technical devices. Tourists usually do not know whether a jellyfish bloom is taking place at the beach of their destination, until they actually arrive. Therefore, providing such information in real time can help them avoid jellyfish-infested beaches. An example of this type of public policy mechanism is the MedJelly application (Marambio et al., 2013), which is made available as a smartphone application by the name of 'iMedJelly'. This application provides daily observations on the status of the Catalan beaches, including information on the presence of the jellyfish outbreaks.

With regard to management of HABs, Anderson (2009) distinguishes prevention, mitigation, and control measures. Prevention entails addressing drivers of HABs, such as nutrient discharge, although much is yet unknown regarding the link between HABs and the biophysical environment. Mitigation, i.e. dealing with an ongoing bloom and its negative impacts, entails the detection of dangerous toxin levels in shellfish, harvesting restrictions, and moving fish pens away from bloom sites. Being able to predict HABs accurately enables authorities to impose mitigation measures on time, thereby enhancing their effectiveness. Jin and Hoagland (2008) investigates the economic value of a prediction system that enables shellfish farmers in the Gulf of Maine to harvest shellfish in advance of closure. If predictions are 100% accurate and blooms occur on average once in two years, the prediction system is worth more than 2005 US$3 million per year in the states of Massachusetts and Maine. This value, however, declines rapidly with lower accuracy and bloom frequency. Lastly, control measures, i.e. measures to limit or reduce the size of a bloom, include mechanical, biological, chemical, genetic and environmental interventions (Anderson, 2009). So far, however, only mechanical measures have been applied on a significant scale, namely dispersal of clay particles to remove harmful algal cells from the water column. Park et al. (2013) estimate that treating 20^5 m^2 (3.2 km^2) with clay particles costs between US$37,700 and US$72,160 per day; the authors indicate that clay dispersal has reduced economic losses due to HABs by more than 80% during earlier blooms.

Table 5: Cost estimates of policy responses cited in this review, with their equivalent in 2010 Purchasing Power Parity dollars

Source	Description	Estimate	Estimate in PPP (constant 2010 international $)
Frésard and Boncoeur, 2006	Costs of controlling *Crepidula fornicata* in the Bay of St-Brieuc, France	€ 4.3 mln - € 6.9 mln	$ 5.64 mln - $ 9.06 mln
Park et al., 2013	Daily costs of treating 20^5 m^2 (3.2	US$ 37,700 -	$ 67,568 -

	km^2) with clay particles to remove harmful algal cells	US$ 72,160	$ 129,329
Fernandes et al., 2016	Costs of regular maintenance and cleaning to avoid hull fouling	5%-10% of annual costs for smaller vessels; 1%-3% for larger vessels	-
Fernandes et al., 2016	Costs of ballast water treatment systems	1.4%-2.9% of annual shipping costs	-

4 Discussion, conclusions, and recommendations for further research

Marine ecological change will likely impact economic well-being in many coastal regions around the world. Economic research to quantify these impacts can facilitate decision-making processes on policy priorities and measures to be taken. In this review we provide an overview of studies on the economic impacts of such changes, particularly those of invasive alien species (IAS), harmful algal blooms (HABs), and gradual changes in species distribution and productivity.

Our review reveals largely three approaches to quantify the economic impact of marine ecological change. The first is to measure the impact of actual events (e.g. Kawahara et al., 2006; Jin et al., 2008), which has the benefit of being based on actual observations, implicitly including some of the behavioural responses by resource users, such as relocation of fishing activities. Behavioural responses that are usually not considered include consumers substituting the affected species by other species. The second approach is to model the ecological relations and economic linkages, ranging from single-species, single-fishery bioeconomic models (see e.g. Knowler, 2005), to multispecies, single-fishery models (see e.g. Bartelings and Hamon, 2015), to multi-sector general equilibrium models like the ICES model (Bosello et al., 2012). This approach can take into account a wide variety of mechanisms, including population dynamics, fisher behaviour, and market responses. The third approach is to conduct surveys among individual residents, tourists, and other recipients of the ecosystem services impacted by marine ecological change. The majority of such approaches estimate individuals' WTP for a reduction of harmful impacts (see e.g. Taylor and Longo, 2010; Börger et al., 2014). This allows for the estimation of non-use values, unlike travel cost studies such as Ghermandi et al. (2015) that focus on use values, i.e. tourism. Survey-based approaches can capture consumer behaviour (hypothetical in choice experiments; actual in travel cost surveys), but the methodological limitations of these studies are well-documented (see e.g. Hausman, 2012).

Of the three changes we consider, invasive alien species and outbreak-forming species have in common that they entail an enhanced abundance of a harmful species or group of species. Indeed, the two overlap to some extent as some invasive species are problematic due to their outbreak-forming nature: some problematic jellyfish, notably *Mnemiopsis leidyi*, are invasive alien species. Gradual changes in species composition and productivity take place on a longer time scale with less empirical evidence to draw on. This makes it more difficult to estimate their impact and requires the use of future projections such as model studies and hypothetical survey questions. Nevertheless, the main driver of gradual changes, namely the changing global climate, may also enhance the other two changes. This illustrates that the three changes discussed in this review should not be considered in isolation, but are mutually dependent.

Our review suggests that the fisheries sector suffers most from impacts on commercial fish species through predation and competition by invasive or outbreak-forming species. This includes exotic species like *Crepidula fornicata* or *Mnemiopsis Leidyi*, but also indigenous species like *P. noctiluca*. For shellfish fisheries harmful algae are most problematic as they can poison shellfish or make them unfit for consumption. Some invasive species, such as *Eriocheir sinensis* or *Crassostrea gigas*, also have a commercial value. The small share of the fisheries sector in most economies suggests that these impacts will be small on a national scale, but as some regions depend heavily on fishing the local impacts can still be serious. The impact of spatial shifts of commercial fish stocks under climate change is likely to be small in comparison with the impact of the possible changes in fuel and food prices.

Tourism appears to be impacted mainly by jellyfish and algae species, and the non-market valuation studies cited in this review suggest that they can have serious impacts. For the invasive shellfish species considered in VECTORS, *Crassostrea gigas*, however, the impact on tourism does not seem to be significant, despite the injuries observed in Dutch beach tourism regions.

Our review also suggests a number of areas where further economic research is warranted. First, although the effects of marine ecological change on the wider economy have been studied earlier (see e.g. Ciscar et al., 2011; Jin et al., 2012), invasive or outbreak-forming species have not yet received much attention in this literature (see e.g. Dyson and Huppert, 2010; Nastav et al., 2013). Second, more economic analyses are needed of prevention, mitigation, and control measures, preferably in a social cost-benefit analysis. This seems especially relevant for jellyfish and harmful algal blooms, which can be controlled by a wide variety of yet underdeveloped measures. Social cost-benefit analysis of generic prevention measures of invasive alien species, such as ballast water management, will be much more difficult because of the wide variety in potentially invasive species, and the difficulty of predicting their impact. Nevertheless, social cost-benefit analysis could help policy-makers in setting priorities in marine policy.

Acknowledgements

The research leading to this publication has received funding from the European Community's Seventh Framework Programme (FP7/2007-2013) within the Ocean of Tomorrow call under Grant Agreement No. 266445 for the project Vectors of Change in Oceans and Seas Marine Life, Impact on Economic Sectors (VECTORS). The authors wish to thank two anonymous reviewers for their helpful comments. Any remaining errors are the sole responsibility of the authors.

Figure captions

Figure 1. Geographical distribution of studies cited in this review.

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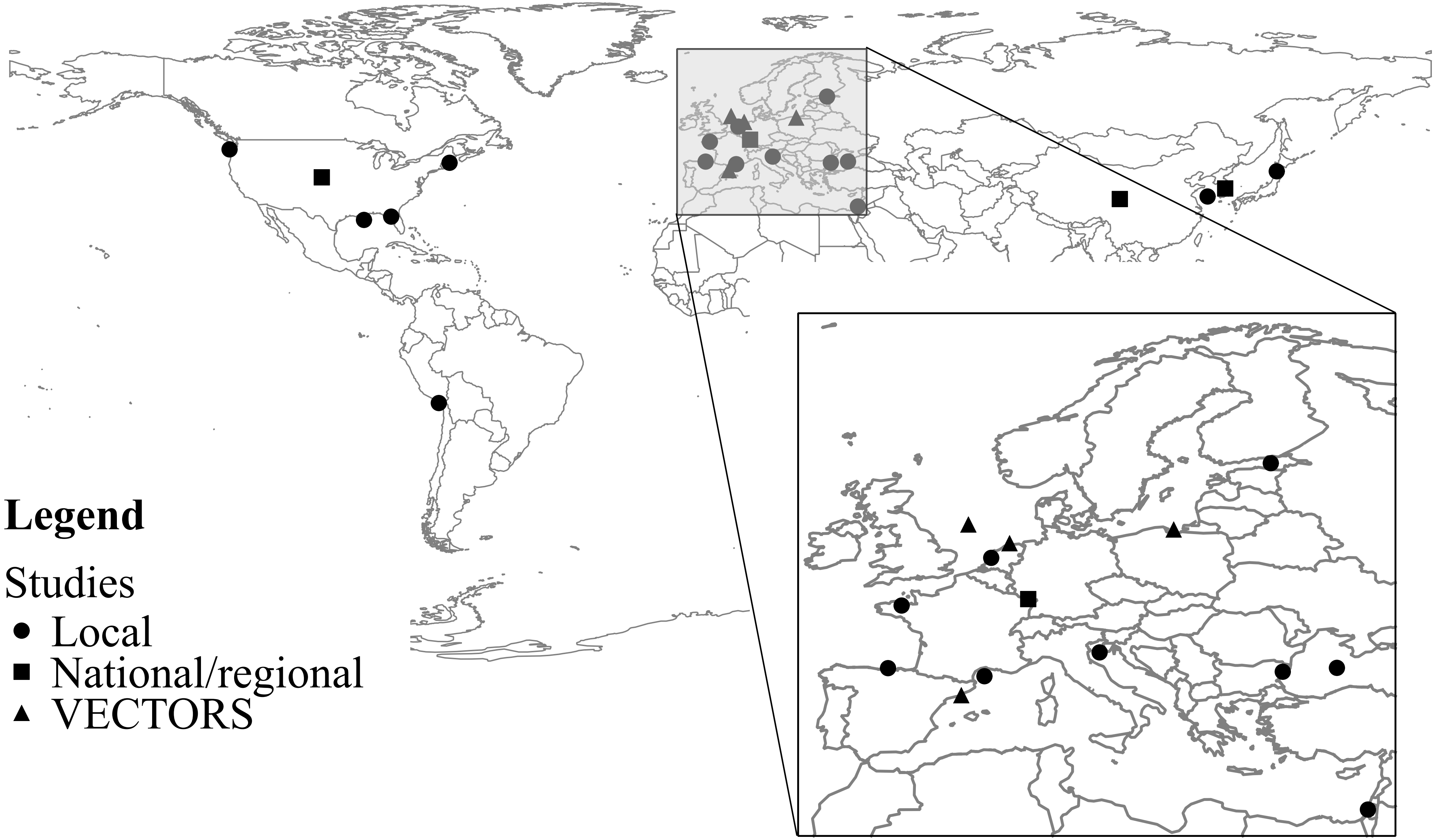

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