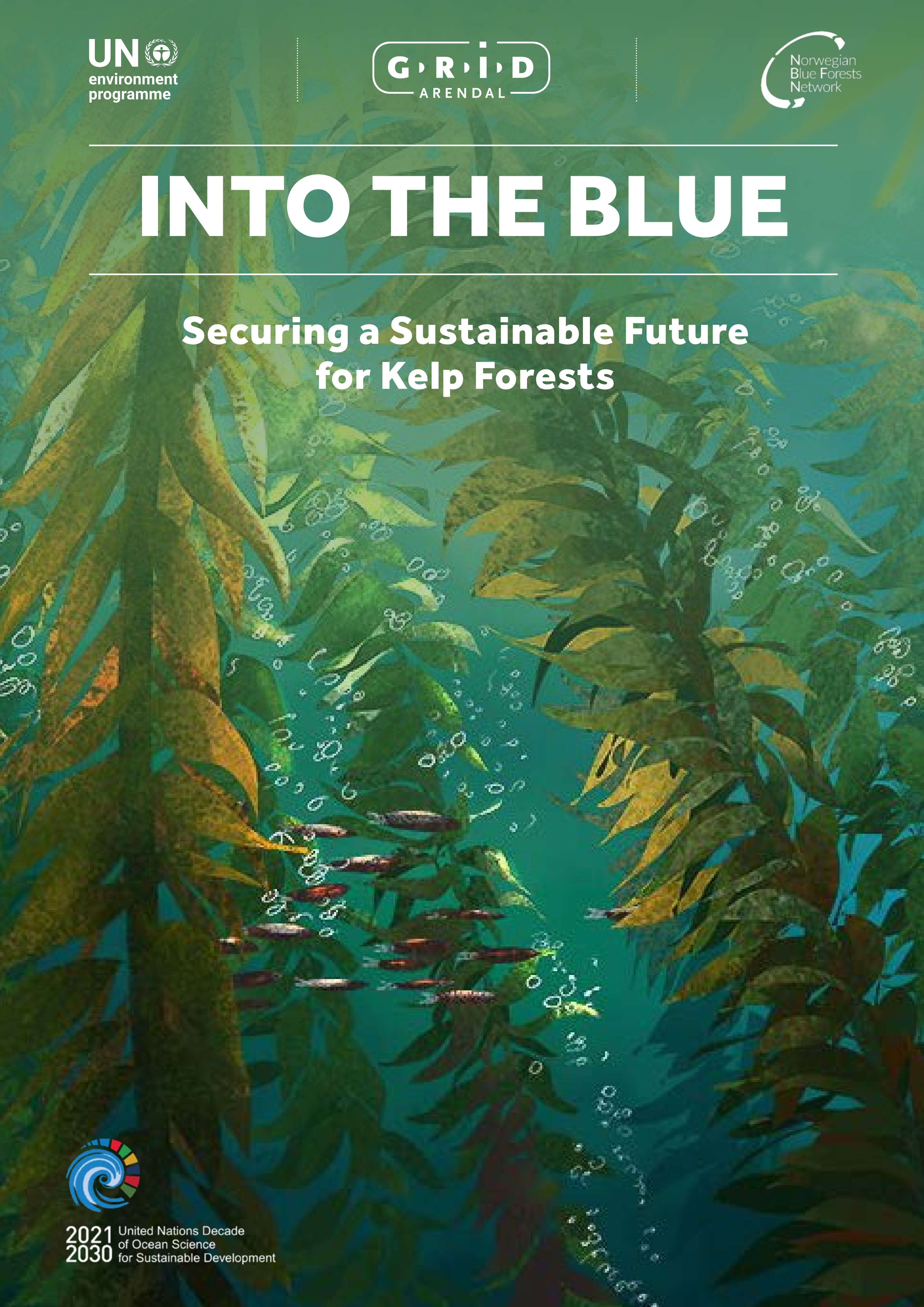

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# INTO THE BLUE

## Securing a Sustainable Future for Kelp Forests

2021 2030 United Nations Decade of Ocean Science for Sustainable Development

# Chapter 2. Status, trends and future projections

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## Highlights

- **Overall, there is a global decline in kelp abundance of 1.8 per cent per year (instantaneous rate).**
- **There is high regional variation in kelp forest status. However, the majority (61 per cent) of longer-term (> 20 years) data sets showed significant declines in kelp abundance, with only 5 per cent showing increases.**
- **Kelp forests at warm range-edges are declining in the North Atlantic, North Pacific and Oceania, often being replaced by mats of turf algae.**
- **Sea urchin overgrazing of kelp forests has occurred across large areas of reef in the Gulf of Maine and eastern Canada, Norway, California, Japan, Korea, Chile, Alaska and Australia.**
- **Kelp forests in Arctic regions are predicted to increase in cover and standing stock with sea ice loss, however increased turbidity may offset these gains in some regions. High Arctic kelp are also predicted to be replaced by temperate kelp.**
- **Most of the world's kelp forests are unmapped and sporadically monitored, making it difficult to detect changes.**

## Status and trends

### Global overview

Kelp forests have declined throughout their temperate range over the past 50 years, as documented by a combination of regional analyses and a global synthesis of available data. The most recent global assessment (Krumhansl *et al*. 2016), mainly representing temperate kelp forests, found an overall global decline in kelp abundance of 1.8 per cent per year (instantaneous rate). This decline occurred amid significant regional variation, with about 38 per cent of ecoregions experiencing declines in kelp abundance, 27 per cent experiencing increases, and the remaining regions showing no detectable change (Krumhansl *et al*. 2016). However, the majority (61 per cent) of longer-term (> 20 years) data sets in the analysis showed significant declines in kelp abundance, with only 5 per cent showing increases (Wernberg *et al*. 2019). This suggests that high short-term variability and the short duration of most kelp forest monitoring data sets included in the Krumhansl *et al*. (2016) analysis may have precluded detections of long-term change.

More recent analyses from continued regional monitoring (e.g. Feehan, Grace and Narvaez 2019; Filbee-Dexter *et al*. 2020; Wernberg *et al*. 2016; Rogers-Bennett and Catton 2019) indicate that declines have persisted and have been exacerbated in some temperate regions, often by ongoing climate-related stress (e.g. warming, marine heatwaves) (Wernberg *et al*. 2016; Arafeh-Dalmau *et al*. 2019; Filbee-Dexter *et al*. 2020) in combination with other stressors (e.g. invasive species, eutrophication, urchin grazing, trophic cascades). See chapter 1 for a detailed description of drivers of change.

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There is evidence of kelp persistence in some temperate regions where climate change stressors have been less intense (Mora-Soto *et al*. 2021; Wernberg *et al*. 2013; Reed *et al*. 2016; Pfister *et al*. 2017) and/or favourable to kelp (Bolton *et al*. 2012). Furthermore, regional variation in environmental conditions and other interacting stressors may result in the persistence of refuge populations. For example, offshore ledges in the Gulf of Maine have been found to contain healthy and thriving kelp compared to degraded inshore reefs, and more exposed headlands that have cooler water temperatures and greater flushing from oceanic processes can support persistent kelp populations in regions where kelp have been lost in more wave-protected areas (Filbee-Dexter, Feehan and Scheibling 2016; Hamilton *et al*. 2020). Some regions have experienced a shift towards kelp species that are better adapted to warm conditions, resulting in little to no overall loss in kelp, but a change in habitat structure and function. Efforts to re-establish healthy kelp populations through mitigation of local stressors and kelp restoration have been successful in some areas (Watson and Estes 2011; Eger *et al*. 2020), though these efforts are expected to be most effective when combined with genetic technologies to address the ongoing and increasing effects of climate change (Coleman *et al*. 2020).

Compared to temperate regions, the trajectory of kelp in Arctic regions is generally much less well known. While poleward range expansions or increased abundance of kelp in the Arctic are predicted with climate change, documented instances of these expansions have been limited (Krause-Jensen *et al*. 2020). In most Arctic regions, limited monitoring has prevented detections of long-term change (see, for example, Merzouk and Johnson 2011), and therefore the status of kelp throughout a large part of their global range remains unknown.


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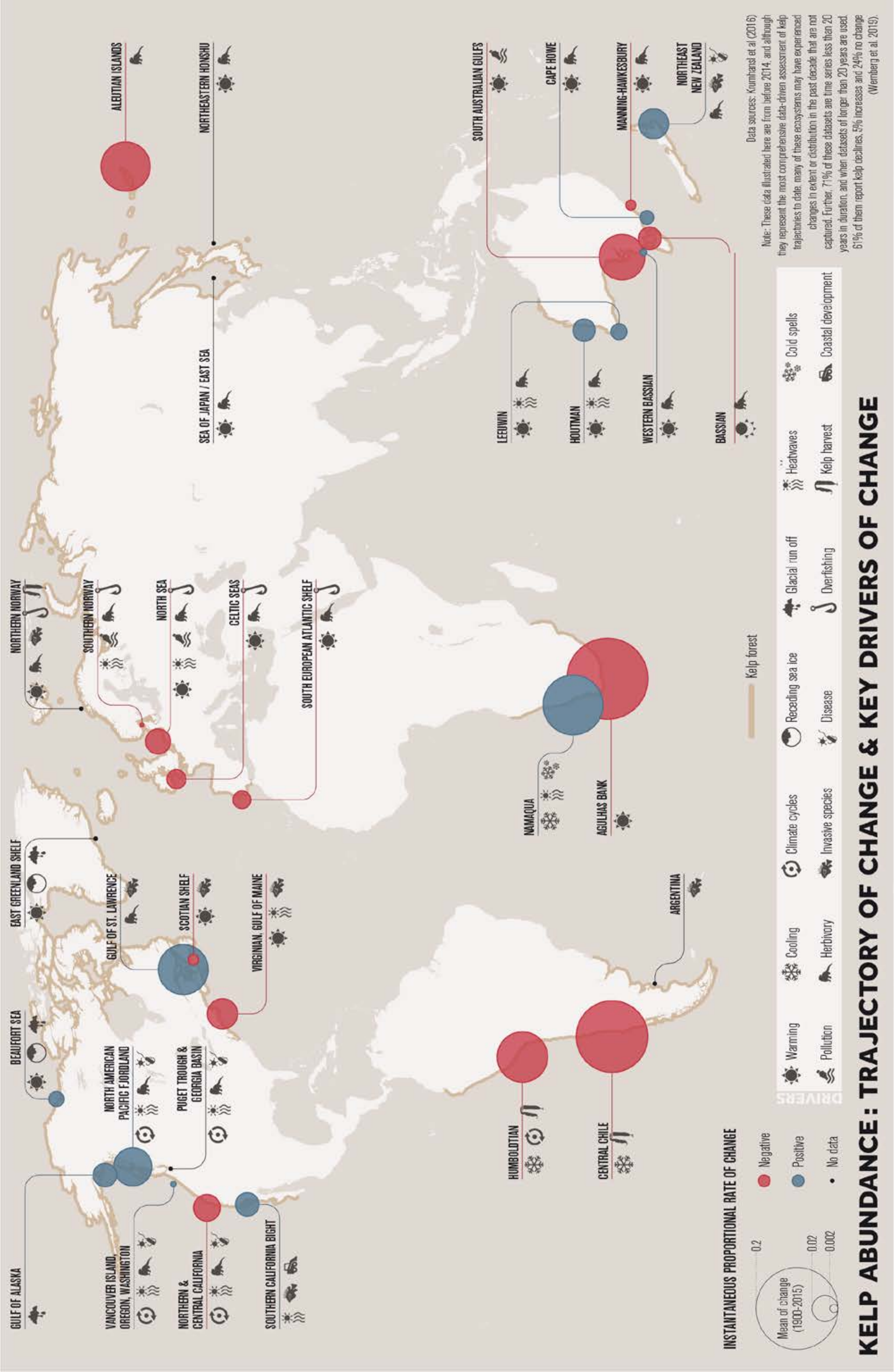
**Figure 2.1.** Trajectory of change in kelp abundance and key drivers of change, by ecoregion globally



Data sources: Krumhansl et al (2016).

Note: These data illustrated here are from before 2014, and although they represent the most comprehensive data-driven assessment of kelp trajectories to date, many of these ecosystems may have experienced changes in extent or distribution in the past decade that are not captured. Further, 71% of these datasets are time series less than 20 years in duration, and when datasets of longer than 20 years are used, 61% of them report kelp declines, 5% increases and 24% no change (Wernberg et al. 2019).

The following sections describe in detail the historical trends in kelp abundance and the main drivers of change, as is currently best known for each region. Following these regional descriptions, we provide information on what has been projected for the future of kelp forests worldwide.

### North-East Pacific – Baja California, Western United States of America, Western Canada

Historical trends in kelp abundance across the North-East Pacific show a large degree of variability. For example, the global analysis of kelp trends from Krumhansl *et al.* (2016) found positive change for the North American Pacific Fjordland (British Columbia), the Gulf of Alaska (2003–2013) and the Southern California Bight (2003–2013), no change for Oregon, Washington and Vancouver Island, and negative change for North-Central California, Southern California Bight (prior to 2002) and the Aleutian Islands. However, trends also vary within each of these ecoregions (Beas-Luna *et al.* 2020). In the Aleutian Islands, a long-term decline in kelp abundance has been well documented (Estes *et al.* 1998). While data are lacking for multiple species across much of the extensive coastline of British Columbia, declines in canopy-forming bull kelp (*N. luetkeana*) were recently reported for sites in the Gulf Islands (Schroeder *et al.* 2020), and substantial changes in subtidal and intertidal kelp species have been documented for two decades on the west coast of Vancouver Island, including overgrazing by sea urchins and loss of kelp during marine heatwaves (Watson and Estes 2011; Starko *et al.* 2019). Northern areas of Haida Gwaii and the West Coast of Vancouver Island in British Columbia have shown kelp persistence since the 1850s, based on an analysis of British Admiralty Charts (Costa *et al.* 2020). Meanwhile in Washington, the abundances of both bull kelp and giant kelp (*M. pyrifera*) have remained relatively stable along the outer coast of the Olympic Peninsula and the Strait of Juan de Fuca (Pfister *et al.* 2017), but substantial losses of bull kelp have been observed in the wave-sheltered areas of the South Puget Sound (Berry *et al.* 2021). There is a large amount of annual to decadal variability in the abundance of bull kelp and giant kelp from California to Oregon, making it difficult to detect directional trends in this region (Bell *et al.* 2020; Hamilton *et al.* 2020). However, beginning in 2014, northern California experienced a sudden and prolonged collapse of bull kelp, which was unprecedented in the 40 years prior (Rogers-Bennett and Catton 2019; McPherson *et al.* 2021). In Baja California, Mexico, giant kelp abundance is also highly variable, with declines occurring following marine heatwaves, including those associated with strong El Niño events (Edwards 2004; Cavanaugh *et al.* 2019).

The primary pressures on kelp abundance and distribution in the North-East Pacific include marine heatwaves and changes in the abundance of starfish and sea otters, which are predators of kelp-grazing sea urchins. In Alaska and Northern British Columbia, local recovery of kelp forests following the return of sea otters has dominated trends over the past century in certain areas (Estes and Palmisano 1974; Watson and Estes 2011). However, areas where sea otters are still absent (such as the Aleutian Islands) have not exhibited similar recovery (Estes *et al.* 1998; Gabara, Konar and Edwards 2021). Furthermore, starfish wasting disease has caused dramatic loss in the population of predatory sunflower starfish across British Columbia, leading to sea urchin outbreaks and subsequent loss of kelp (Burt *et al.* 2018). Along the coast of Washington, fluctuations in kelp abundance have been linked to climate cycles such as the Pacific Decadal Oscillation (PDO), the El Niño-Southern Oscillation (ENSO) and the North Pacific Gyre Oscillation (NPGO) (Pfister *et al.* 2017). In Northern California, the sudden collapse in bull kelp populations has been linked to a marine heatwave and a boom in sea urchin recruitment following outbreaks of starfish wasting disease (Rogers-Bennett and Catton 2019; McPherson *et al.* 2021). In Southern California and Baja Mexico, heatwaves and climate anomalies have led to temporary declines in kelp abundance, with a large degree of spatial variability in patterns of recovery (Arafeh-Dalmau *et al.* 2019; Cavanaugh *et al.* 2019). On local scales, non-indigenous tropical algae (e.g. *S. horneri, S. muticum* and *U. pinnatifida*) has invaded kelp forests at sites across southern California (Marks *et al.* 2015) and sedimentation from coastal development has been linked to declines in kelp forest abundance at localized sites in Southern California and Baja Mexico (Foster and Schiel 2010; Torres-Moye and Escofet 2014).

### South America – Argentina, Chile and Peru

In recent decades, there has been a significant decrease in the abundance of kelp in northern and central Chile, despite a cooling trend in this region (Krumhansl *et al.* 2016; Vásquez 2008). Kelp abundance has been more stable in southern Chile and the South Georgia and Falkland Islands (Malvinas), an area that accounts for more than 47 per cent of the known distribution of giant kelp, *M. pyrifera* (Mora-Soto *et al.* 2020). A comparison of satellite imagery from 1984–2019 with nautical charts and surveys from the nineteenth and twentieth centuries found that most kelp forests in this region were remarkably persistent (Mora-Soto *et al.* 2021). Another study of giant kelp dynamics on the southern tip of South America found no significant change in kelp abundance and associated biodiversity over the past 45 years (Friedlander *et al.* 2020), but further north along the Argentinean coast the introduction of wakame (*U. pinnatifida*) already covers 1,850 km of the Atlantic coastline (Bunicontro, Marcomini and Casas 2018). The stands of this invasive kelp reduce native seaweed diversity (Casas, Scrosati and Piriz 2004) and can also invade giant kelp forests (Raffo, Eyras and Iribarne 2009) and modify the habitat for coastal fish (Irigoyen, Eyras and Parma 2011).

Decreases in the abundance of kelp in northern and central Chile are likely related to high extraction pressure, as this region has the highest rate of natural kelp harvest in the world (Buschmann *et al.* 2014; Camus, del Carmen

Hernández-González and Buschmann 2019; Vega *et al.* 2019). High harvest rates have also led to changes in the population structure and morphology of some kelp species (e.g. *L. berteroana, L. trabeculata*), with high harvest intensities being associated with increases in juvenile densities and smaller kelp size (Vega, Broitman and Vásquez 2014; Gouraguine *et al.* 2021). Nevertheless, territorial user rights policies and the development of strategies for marine protected areas by the Chilean Government have contributed to the conservation of wild intertidal *Lessonia* populations (González-Roca *et al.* 2021).

In southern Chile, kelp populations seem more stable, are much less impacted by direct human pressures and may not yet have experienced warming sufficient to cause regional declines in kelp abundance (Krumhansl *et al.* 2016; Mora-Soto *et al.* 2020). Moreover, recent evidence suggests that kelp (e.g. giant kelp) are adapting to the increased presence of fresh water and sediment from melting glaciers (Palacios *et al.* 2021) and that despite there being no inter-population differences in response to ocean acidification and warming, intrinsic differences exist among populations that seem to be associated with their natural variability in $CO_2$, $NO_3$ and seawater temperatures driven by coastal upwelling (Fernández *et al.* 2021). A northern shift in the polar front, intense westerlies and the cooling effect of glacier melting would keep temperatures in the normal range for kelp tolerance (Mora-Soto *et al.* 2022).

### North-West Atlantic – Eastern United States of America, Nova Scotia, Gulf of St. Lawrence, and Newfoundland

Studies of long-term change in the North-West Atlantic region (1952–2014) have documented widespread and significant declines in kelp abundances (*S. latissima, L. digitata* and *Agarum cribosum*), particularly throughout the southern portion of the range, including Southern New England, the Gulf of Maine, and along the Atlantic coast of Nova Scotia (Filbee-Dexter, Feehan and Scheibling 2016; Feehan, Grace and Narvaez 2019; Filbee-Dexter *et al.* 2020). Although kelp-dominated sites remain in these parts of the region, many kelp forests have given way to carpets of turfs made up of many species of native and invasive seaweed (Pessarrodona *et al.* 2021). The loss of kelp forests from their southern range-edge (Southern New England, United States of America) is correlated with an increase in the cumulative annual intensity of marine heatwaves that have equalled or exceeded a 22.8°C thermal threshold for sugar kelp (*S. latissima*) mortality (Filbee-Dexter *et al.* 2020). The loss of kelp forests throughout the North-West Atlantic is also linked to the direct and indirect effects of rapid multidecadal ocean warming (Krumhansl, Lee and Scheibling 2011; Filbee-Dexter, Feehan and Scheibling 2016; Witman and Lamb 2018; Dijkstra *et al.* 2019). Warm temperatures are associated with reduced kelp growth rates, high mortality, and increased tissue loss associated with temperature-mediated interactions with small grazers and encrusting species, and the subsequent proliferation of algal turfs (Simonsen, Scheibling and Metaxas 2015a and 2015b; Krumhansl, Lee and Scheibling 2011; Dijkstra *et al.* 2019).

Historically, sea urchins were the main driver of kelp abundances through time in the North-West Atlantic, causing widespread barrens that persisted for years to decades along hundreds of kilometres of coastline (Filbee-Dexter and Scheibling 2014). From the late 1960s to the 2000s, the Atlantic coast of Nova Scotia (Canada) underwent decadal fluctuations between kelp-forested and urchin barren states, driven by recurrent disease-induced sea urchin (*Strongylocentrotus droebachiensis*) mass mortalities and subsequent sea urchin population recovery. Over the last decade, kelp forests have shifted to an algal turf state following the decimation of sea urchins by intensifying disease and colonization of turfs under warming sea temperatures. Beginning in the 1970s and 1980s, sea urchins destructively grazed kelp forests in the Gulf of Maine (Harris and Tyrrell 2001; Steneck, Vavrinec and Leland 2004), but their barrens collapsed in the early 1990s following the opening of a sea urchin fishery that decimated urchin populations (Taylor 2004; Steneck, Vavrinec and Leland 2004). Kelp declines have not been documented in the Gulf of St. Lawrence, Newfoundland, and Labrador, yet data from much of these areas are sparse. Although sea urchin barrens are prevalent in these areas, it is unclear if the areas were historically dominated by kelp forests that were overgrazed in the past century or if they have been persistent barrens throughout recent history (Gagnon, Himmelman and Johnson 2004; Merzouk and Johnson 2011; Frey and Gagnon 2016; Krumhansl *et al.* 2016).

### North-East Atlantic – Europe, UK

Studies from the North-East Atlantic have reported variable trends with respect to kelp abundances, with documented increases and decreases over the past few decades (Krumhansl *et al.* 2016). *S. polyschides* is a native kelp species that follows the general trend of decline along its southern marginal range in Europe (Araújo *et al.* 2016). Reduction in abundance, local extinctions or range contractions for kelp have been reported in southern Norway (sugar kelp, Filbee-Dexter *et al.* 2020), Spain and Portugal (*L. hyperborea* and *S. polyschides*, Smale 2020), often associated with shifts in dominance towards algal turfs. *L. ochroleuca* has reduced in abundance in the Bay of Biscay, Spain (Araújo *et al.* 2016) but is increasing along its northern range-edge in the south-west of the UK (Smale 2020). *L. digitata* is experiencing a range shift, with declines along the southern range-edge in the English Channel (King *et al.* 2020). *L. hyperborea* has expanded in Helgoland, Germany, where it is displacing *S. latissima* and *L. digitata* (Araújo *et al.* 2016). Along the northernmost part of the Norwegian coast, kelp forests have largely been reduced, though both *Laminaria* and sugar kelp forests are recovering in mid-Norway in part of a 2,000 $km^2$ area that was formerly dominated by sea urchins. This recovery, starting in the 1990s, is still progressing at the southern barrens' limit and to a lesser extent around the northern barrens' limit, more than five decades after sea urchins bloomed and overgrazed the kelp forests (Christie *et al*. 2019a).

Warming and overfishing are the main drivers of kelp loss and changes in kelp distribution in the North-East Atlantic. The gradual temperature increase in coastal waters in this region has caused northward range shifts for many kelp species (Smale 2020), and the broad-scale mortality of kelp in marginal areas (e.g. sugar kelp in the Skagerrak, southern Norway) has been linked to increasingly frequent marine heatwaves (Filbee-Dexter *et al*. 2020). A sea urchin bloom on the mid- and northern Norway coast occurred around 1970 and was most likely caused by overfishing and a collapse in coastal fish stocks, leading to the release of a prominent grazer, the sea urchin *Strongylocentrotus droebachiensis* (Norderhaug *et al*. 2020). The ongoing reduction of sea urchins and recovery of kelp are consequences of gradual warming and expansion of predatory crabs (Christie *et al*. 2019b). Although sea urchin grazing is mainly reported from northern coasts in Norway, small-scale localized events have been reported widely from France, the UK and Denmark in the south to Iceland and the Russian Federation in the north (Norderhaug and Christie 2009). Other causes of kelp loss include eutrophication and freshwater run-off from land (Bartsch *et al*. 2008; Araújo *et al*. 2016; Filbee-Dexter and Wernberg 2018). Expansion of invasive species and subsequent displacement of native species is also a threat at present and will be in the future. For example, *U. pinnatifida* (wakame) was introduced to the Mediterranean Sea from Asia in the 1970s and is now expanding along the Iberian and French coast (Araújo *et al*. 2016). Finally, the growing interest in kelp cultivation in Europe poses a potential future environmental threat by increasing the risk of spreading non-native species and genes (Campbell *et al*. 2019).

## Southern Africa

Cooling sea surface temperatures in western South Africa have been associated with increases in kelp abundance and kelp range expansion. While sea surface temperatures have been rising throughout much of the world, in this region increases in south-easterly winds and upwelling beginning in the 1990s have resulted in decreased annual mean temperatures (Blamey *et al*. 2015). In the mid-1990s, a floating canopy of *E. maxima* appeared at De Hoop Nature Reserve, approximately 70 km east of the documented long-term range limit of this species (Bolton *et al*. 2012). This population has persisted, but the species is still not reported at sites between the De Hoop Nature Reserve and the previous long-term range limit (John J. Bolton, pers. obs., 2007–2020). In contrast, climate change is resulting in warming and strengthening of the Agulhas Current, the major warm surface current moving southward along the east coast of South Africa (Blamey *et al*. 2015). These changes have not yet resulted in significant changes in kelp abundance, but future warming may affect the small inshore populations of *E. radiata* in this region.

Notably, there have been major changes in kelp forest fauna in South Africa since the 1970s, with almost all populations of reef fish species overexploited or collapsed, and both West Coast rock lobster and abalone populations severely overexploited (Blamey and Bolton 2018). Unlike many other regions, these faunal changes have not yet led to loss of kelp forests in South Africa, although they have been linked to other changes in kelp forest community structure. For example, the presence of sea urchins along the Cape of Good Hope has corresponded to changes in kelp forest composition, but not changes in overall kelp abundance (Leliaert *et al*. 2000). Additionally, an eastward migration of West Coast rock lobsters in the 1990s into a portion of the urchin-dominated coastline removed sea urchins and resulted in changes to the kelp forest understorey community (Blamey and Branch 2012).

## North-West Pacific – Republic of Korea and Japan

Decadal scale warming trends in the North-West Pacific have been linked to range contractions of temperate kelp species such as *E. cava* and expansion of some warm-water-tolerant kelp and coral species. In south-western Japan, warming during the 1997–1998 ENSO was associated with a large decline in E. cava and a persistent expansion of tropical species (Tanaka *et al*. 2012). Declines in kelp abundance in other parts of Japan have been associated with grazing from herbivorous fish, which warming can increase (Nakayama and Arai 1999; Masuda *et al*. 2000). In the Cape Ōma region of northern Japan, seawater temperature increases of around 1°C between the 1980s and early 2000s corresponded to a decrease in the abundance of cold-temperate species including *S. japonica* (kombu) and increases in warm species such as *U. pinnatifida* (wakame) (Kirihara *et al*. 2006).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Warming has also been linked to range contractions of temperate species in Jeju Island, Republic of Korea (Kang and Chung 2015). *E. cava*, the dominant kelp species in this area, share shallow benthic habitats with soft coral species and dynamically fluctuate in response to various disturbances, including summer typhoons (Kim *et al*. 2021). There are also declines in kelp abundance and shifts towards urchin barren states along the east coast of the Republic of Korea (Jeon, Yang and Kim 2015; Hong *et al*. 2021). A recent study demonstrated that removal of sea urchins led to rapid kelp recovery as well as the stability of the food web structure, suggesting that kelp could persist in this region if grazers were reduced (Hong *et al*. 2021; Kim *et al*. 2022).

### Oceania – Australia and New Zealand

South-eastern and south-western Australia have warmed faster than 90 per cent of the global ocean (Hobday and Pecl 2014). In response, these regions have experienced a poleward contraction of temperate seaweed species over the past 50 years (Wernberg *et al*. 2011a). In 2011, the west coast of Western Australia lost 43 per cent of its *E. radiata* (golden kelp) forests (Wernberg *et al*. 2016), and other habitat-forming fucoids. The marine heatwave caused an approximately 100 km range contraction of kelp forests and substantial tropicalization of associated reef communities (Wernberg *et al*. 2016). These impacts were compounded by increased grazing from range-shifting tropical fish (Bennett *et al*. 2015; Zarco-Perello *et al*. 2017). Now, more than 10 years later, these kelp forests have not recovered (Wernberg 2020). Further to the south, where the heatwave was equally intense but did not reach temperatures in excess of the thermal limit of golden kelp, forests showed little response (Wernberg *et al*. 2013).

Losses of golden kelp have also occurred in south Australia in the Adelaide gulfs and in Port Phillip Bay, Victoria (Carnell and Keough 2019) due to eutrophication and warming, and in northern New South Wales due to grazing from range-shifting tropical herbivores (Vergés *et al*. 2016). Around Tasmania, giant kelp has declined by more than 95 per cent in the past few decades, mainly due to warming, incursions of nutrient-poor warm water and grazing from range-shifting sea urchins (Johnson *et al*. 2011; Ling and Keane 2018). In New Zealand, golden kelp (*E. radiata*) has experienced local die-offs and reductions in abundance along the North Island due to diseases and grazing (Cole and Babcock 1996; Haggitt and Babcock 2003; Shears, Babcock and Salomon 2008), while giant kelp has been lost in Otago due to increased turbidity, heatwaves and urchin grazing (Glover 2021). The invasive kelp wakame has invaded some parts of the region and can outcompete some native kelp species. In 2017/18, populations of *D. poha* were lost during a marine heatwave near Lyttelton Harbour on the South Island (Thomsen *et al*. 2019).

### Arctic – United States of America, Canada, Greenland, Svalbard and the Russian Federation

Arctic kelp forests have been vastly understudied relative to kelp in temperate regions (Krumhansl *et al*. 2016), but recent studies demonstrate widespread and extensive kelp coverage in the Arctic. The most recent pan-Arctic review of existing long-term monitoring and field studies on kelp ecosystems shows a general trend of increasing abundance in response to climate change (Krause-Jensen *et al*. 2020), though trends are variable across existing studies that cover some parts of the Beaufort Sea, Greenland, Norway and the Russian Federation. For example, a 40-year time series in the Beaufort Sea showed no change in kelp productivity or abundance (Krumhansl *et al*. 2016; Bonsell and Dunton 2018), while diver surveys in Arctic Norway (Svalbard) reported an eightfold increase in kelp biomass (*L. digitata*) in 2012–2014 compared to 1996–1998 (Bartsch *et al*. 2016). However, this was also associated with a loss of kelp biomass in deeper areas, which was attributed to reduced light from higher turbidity due to climate-driven increases in terrestrial run-off (Pavlov *et al*. 2019). In the Russian Federation, the recovery of kelp in sea urchin barrens has occurred in the Guba Zelenaya Fjord, but kelp forests have retreated from lower depth limits of 10–12 m in Kola Bay near Murmansk. There is also some evidence that *L. hyperborea* is extending its range east, further into the Russian Federation as temperatures increase and sea ice retreats (Krause-Jensen *et al*. 2020).

Historical changes in Arctic kelp forests have been closely linked to environmental conditions, with sea ice cover being one of the primary drivers of change. Sea ice cover has become thinner and has begun to break up earlier in the year, reaching historic lows in the last decade. As sea ice restricts the upper depth limits of kelp by mechanical abrasion and restricts the lower depth limits by light shading (Filbee-Dexter *et al*. 2019), its loss generally makes habitat more suitable for kelp. However, the coastal zone is also the main recipient of increasing sediment fluxes from thawing permafrost and eroding continental shelves (Fritz, Vonk and Lantuit 2017), as well as freshwater inputs from increased river discharge and glacier melt (Meredith *et al*. 2020). Increased freshwater run-off from glacial ice melt increases turbidity in coastal waters, which can lead to kelp shading (Bonsell and Dunton 2018; Pavlov *et al*. 2019) and offset the beneficial effects of decreasing ice cover for kelp populations. The degree to which these environmental changes have impacted kelp abundances varies regionally and depends on the extent to which melting sea ice, glacial melt, and permafrost erosion increase turbidity and freshening in coastal areas (Bartsch *et al*. 2016; Traiger and Konar 2018).

## Future projections

During the remainder of the twenty-first century, anthropogenic climate change is expected to cause further increases in mean ocean temperatures and the frequency and intensity of marine heatwaves, reductions in sea ice cover in the Arctic and Antarctic, and increased glacier melt and run-off at high latitudes (Frölicher *et al*. 2018; Oliver *et al*. 2019). Increases in the uptake of atmospheric $CO_2$ by surface waters will result in increased partial pressure of carbon dioxide ($pCO_2$) levels and ocean acidification (Doney *et al*. 2009). Climate change is also expected to increase storm frequency and intensity in some regions, leading to higher levels of wave disturbance (Reguero, Losada and Méndez 2019).

In general, climate change is likely to shift kelp distributions poleward, with declines in equatorward range-edge populations and expansions at poleward range-edges (Table 2.1), a pattern that corresponds to recent observed trends in kelp abundance (Smale 2020). Warm-tolerant species are expected to replace cold water species, thus altering the diversity and community composition of kelp forests (Assis, Araújo and Serrão 2018; Sudo *et al*. 2020). In polar latitudes, reductions in sea ice will lead to lower salinity and greater sediment inputs in kelp ecosystems (Filbee-Dexter *et al*. 2019). This pattern is prevalent in the North Atlantic, which is probably the most well-studied region in terms of future projections. In this region, contractions in warm range-edges and northward shifts have been projected for numerous species (Assis, Araújo and Serrão 2018). In Australia, trends of decline in abundance and distribution are expected to continue, leading to the extinction of some species such as *M. pyrifera* and major range contractions of other iconic species, including *E. radiata* (Martínez *et al*. 2018). The Aleutian Islands are another region of particular concern as both kelp forests and the biogenic coralline habitat on which they are built are declining due to herbivory, warming temperatures and ocean acidification (Rasher *et al*. 2020).

However, there will be local to regional scale variability in how kelp forests respond to climate change. Local adaption (e.g. Vranken *et al*. 2021) and intraspecific variability in tolerance to warming (e.g. Clark *et al*. 2013) may lead to variability in the magnitude of temperature-driven declines across species' ranges (Bennett *et al*. 2015; King *et al*. 2019). These processes may also lead to populations that are particularly resilient due to their adaptation to warm conditions (Lind and Konar 2017). Local variation in ocean currents, upwelling, and bathymetry can create refugia for kelp against climate change (Davis, Champion and Coleman 2021). Changes in kelp forest community composition could alter species interactions and, in some cases, maintain ecosystem functions.

Extrapolating recent trends in kelp abundance may not provide an accurate view of the future because of non-linear responses and thresholds, complex interactions among multiple stressors (see chapter 1), and changing policies related to fishing, coastal development, and climate change mitigation. Although the majority of research focuses on climate-driven impacts, future projections for the world's coastal zones show increases of multiple other important stressors, including heavily intensified food production in these regions (wild harvest and aquaculture), growing coastal development, increased shipping activity and increased pollution and run-off. These changes have all been linked to kelp loss in various regions, yet the exact consequences of these threats are not well understood, and there are limited predictive models to show how these stressors may interact to alter kelp forests in the coming decades.


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**Table 2.1.** Examples of the projected impacts of climate change on the abundance and distributions of major kelp species

| Region | Species | Response | References |
|---|---|---|---|
| Japan | *Alaria crassifolia, Agarum clathratum, Costaria costata, Arthrothamnus bifidus,* seven species of *Saccharina* | Declines in distribution of all species by the 2040s and 2090s, depending on the warming scenario<br>Northern range shifts for most species | Sudo *et al.* 2020 |
| North Atlantic | *Laminaria solidungula, A. esculenta, Saccorhiza dermatodea, L. digitata, L. hyperborea, Saccharina latissima, S. polyschides, L. ochroleuca* | Contraction of low latitude ranges with northwards expansion<br>Under Representative Concentration Pathway (RCP) 8.5 (high-emissions warming scenario), decreases in *L. solidungula, A. esculenta, S. latissima, L. digitata, L. hyperborean* and increases in *S. dermatodea, S. polyschides, L. ochroleuca* | Assis, Araújo and Serrão 2018 |
| North Atlantic | *L. digitata, S. latissima* | Poleward range contractions between 100 and 300 km | Khan *et al.* 2018 |
| Europe | *L. digitata* | Declines in abundance at southern edge with poleward shift of distribution | Raybaud *et al.* 2013 |
| Europe | *L. ochroleuca* | Declines in abundance at southern edge (Morocco and Iberian Peninsula) with poleward shift of distribution (e.g. UK) | Franco *et al.* 2018 |
| Eastern Australia | *Ecklonia radiata* | Poleward range contraction of around 530 km by 2100 | Castro *et al.* 2020 |
| Australia | *Macrocystis pyrifera, E. radiata* | *Macrocystis* projected to become extinct from Australia by 2100 under RCP 6.0<br>*Ecklonia* predicted to contract by between 49 and 71 per cent and become restricted to south coast of Australia | Martínez *et al.* 2018 |
| Eastern North America | *L. digitata, S. latissima* | Expansion at poleward limits, contractions at southern range-edges | Wilson *et al.* 2018 |
| North-East Pacific | *Eualaria fistulosa, Nereocystis luetkeana, M. pyrifera, Eisenia arborea* | Declines in the abundance of kelp near the equatorward range limits of *M. pyrifera* in Baja California and *N. luetkeana* in Northern California<br>Continued decline in Aleutian kelp forest abundance and degradation of biogenic habitat | Beas-Luna *et al.* 2020<br>Rasher *et al.* 2020 |
| Arctic | *L. solidungula, Alaria* spp. *S. latissima* | Severe southern range contractions in endemic Arctic kelp taxa, with up to 67 per cent of suitable habitat lost by 2100 under RCP 8.5. Areas likely to experience losses regardless of climate change severity include coastlines in the Sea of Okhotsk, the Bering Sea and the White Sea. Increasing climate change severity will also result in severe losses in Hudson Bay, Newfoundland and Labrador (Canada), Svalbard (Norway), the Barents Sea (the Russian Federation) and the northernmost reaches of the Bering Sea. Arctic habitat is projected to remain stable along much of the Siberian coastline, the northern half of Greenland, and the Canadian Arctic Archipelago. In areas with projected losses in Arctic taxa, succession by temperate kelp is predicted. | Bringloe *et al.* in 2022 |

## Knowledge gaps

The most comprehensive study to date of historical trends in kelp abundance contained data from only about one third (34 out of 99) of the world's ecoregions where kelp forests exist, and in most cases coverage within these regions was relatively sparse in space and time (Krumhansl *et al*. 2016). Most studies using these data were relatively short in duration, which may contribute to a lack of detection of directional change in many regions, as kelp forests are known to be highly variable on short timescales (Wernberg *et al*. 2019). A lack of consistent funding for monitoring has contributed to a paucity of long-term studies on kelp abundances. Funding lapses for monitoring that lead to data gaps and shifts in methodologies make detecting long-term changes difficult. The lack of data from Arctic regions is of particular concern, and consequently there is a high degree of uncertainty in the trajectory of kelp forests over a large part of the global range of kelp.

A coordinated network of global kelp observations would help address these gaps (e.g. Duffy *et al*. 2019). Remote sensing also represents a valuable tool for monitoring certain species of kelp (e.g. those that form surface canopies), especially in remote areas. Satellite imagery has been used to map giant kelp distributions on global scales (Mora-Soto *et al*. 2020) and characterize trends and variability for a handful of regions such as California, Tasmania and the Falkland Islands (Malvinas) (Bell *et al*. 2020; Butler *et al*. 2020; Houskeeper *et al*. 2022). These efforts should be expanded to map trends on global scales with increasing spatial and temporal resolution.

We also need to better understand the drivers of changes in kelp abundance, especially with respect to multiple interacting pressures. For example, kelp forests in southern California exhibited relatively high resilience to a major marine heatwave in 2014–2016 (Cavanaugh *et al*. 2019), but this same event was linked to collapse of kelp forests in northern California due to interactions between the heatwave and high levels of urchin grazing (Rogers-Bennett and Catton 2019). Understanding interactions between climate change and local pressures such as pollution, overfishing and invasive species is critical for informing management actions. Local pressures can often be mitigated by actions such as regulating coastal development and pollution, implementing marine protected areas, reducing grazing pressure and making restoration/afforestation efforts (Eger *et al*. 2020; Morris *et al*. 2020; Hamilton *et al*. 2022).

While projections of changes in kelp forest distribution have been developed for some species and regions (Table 2.1), there are major geographic and conceptual gaps in our understanding. To date, most projections have focused on the North Atlantic and Europe. There is a lack of studies for widespread surface-canopy forming species such as *M. pyrifera, N. luetkeana* and *E. maxima*, and even fewer standardized studies on subcanopy kelp dynamics. More information is needed for the Eastern and Western Pacific, including areas such as Asia (where there is high demand for harvested kelp) and the North-East Pacific (where commercial interests in seaweed harvesting are currently being developed).

Future modelling efforts should attempt to develop dynamic predictive models that incorporate adaptation, ecological interactions, multiple stressors and extreme events. To achieve this, more data are needed on intraspecific variability in environmental tolerance and variability in the tolerance of different life stages. It would also be useful to develop models that integrate experimental data on species tolerance thresholds to better characterize non-linear responses (e.g. Franco *et al*. 2018; Castro *et al*. 2020). Improved models will enable more accurate projections and give managers the tools to examine the effects of mitigating local stressors to promote kelp forest resilience in the face of climate change.

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