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Opportunities for blue carbon strategies in China

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A R T I C L E I N F O

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A B S T R A C T

Blue Carbon (BC) strategy refers to the approaches that mitigate and adapt to climate change through the conservation and restoration of seagrass, saltmarsh and mangrove ecosystems and, in some BC programs, also through the expansion of seaweed aquaculture. The major losses of coastal habitats in combination with the commitments of China under the Paris Agreement provide unique opportunity and necessity to develop a strong Chinese BC program. Here, we (1) characterize China's BC habitats, examine their changes since 1950 along with the drivers of changes; (2) consider the expansion of seaweed aquaculture and how this may be managed to become an emerging BC resource in China, along with the engineering solutions required to enhance its potential; and (3) provide the rationale and elements for BC program in China. We find China currently has 1326–2149 km^2 wild and 2–15 km^2 created mangrove, saltmarsh and seagrass habitats, while 9236–10059 km^2 (77–87%) has been lost since 1950, mainly due to land reclamation. The current area of farmed seaweed habitat is 1252–1265 km^2, which is close to the area of wild mangrove, saltmarsh and seagrass habitats. We conclude that BC strategies have potentials yet to be fully developed in China, particularly through climate change adaptation benefits such as coastal protection and eco-environmental co-benefits of seaweed farming such as habitat creation for fish and other biota, alleviation of eutrophication, hypoxia and acidification, and the generation of direct and value added products with lower environmental impact relative to land-based production. On this basis, we provide a roadmap for BC strategies adjusted to the unique characteristics and capacities of China.

1. Introduction

China, which currently represents 22% of the world population and accounts for 12% of global emissions of green-house gases and black carbon (Kong et al., 2016), has adopted the Paris Agreement, pledging to reduce the rate of growth in carbon emissions and to reach the peak of

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CO_2 emissions by 2030. Because of its large share of global emissions, the commitment by China is a key to the success of the Paris Agreement. However, achieving the goal of reduced emissions while pursuing its socio-economic development poses a great challenge to China, which requires using the broadest possible range of options to reduce and avoid emissions, while also adapting to climate change. Indeed, China is particularly vulnerable to sea level rise with a large proportion of its population at risk in coastal megacities which have already experienced severe losses due to typhoons and floods (Guan et al., 2015; Nicholls et al., 2007). For instance, super typhoon"Meranti"that made landfall in Xiamen, Fujian province on September 15, 2016, causing direct economic losses of 10.2 billion RMB, severe damage to natural ecological system of the whole city, killing 48 and injuring another 49 persons. From 2012 to 2017, the direct economic losses caused by storm surges in China have exceeded 9.8 billion RMB annually, which accounts for more than 91% of the total direct economic losses by all marine disasters. As China is still a developing nation, with a large population in need additional energy production for further development, some of the options being considered by wealthy and more developed nations cannot form the underpinnings of the Chinese strategy to fulfill their commitment to mitigate climate change. When considering the range of options to be included in national policies, preferred mitigation options should include those that are readily actionable, cost-effective and generate co-benefits, in terms of climate-change adaptation and added value towards other national priorities, while delivering the national commitments under the Paris Agreement.

Blue Carbon (BC) strategies, referring to approaches to mitigate and adapt to climate change based on conservation and restoration of vegetated coastal habitats such as seagrasses, saltmarshes, and mangroves (Duarte et al., 2013; Mcleod et al., 2011; Nellemann et al., 2009), have been recently adopted by many coastal nations as cost-effective strategies that are particularly suitable for developing nations with extensive coastlines (van Kleef et al., 2016). In addition, mangroves, saltmarshes, and seagrass meadows can significantly attenuate wave energy and raise the seafloor, thus protecting the shoreline from sea level rise and erosion (Duarte et al., 2013; Kirwan and Megonigal, 2013). Coastal areas with marine vegetation form important natural buffers against typhoon and wave destruction, flooding erosion of farmland and wetland. Kelp forests have similar capabilities, for example, Norwegian kelp forests have been found to reduce wave heights by up to 60% (Mork, 1996). Farmed seaweeds, prominent in China, could play similar roles, helping protect coastal land from flooding and erosion while contributing to avoid greenhouse gas emissions (Duarte et al., 2017).

Vegetated coastal ecosystems (mainly mangroves, saltmarshes and seagrasses) are particularly effective at capturing CO_2 from the atmosphere and storing them in the roots and soils/sediments as blue carbon sinks, which could contribute to global greenhouse gas emission mitigation (e.g. Duarte et al., 2013; Gattuso et al., 2018). These blue carbon habitats have experienced major losses globally, thereby offering opportunities to restore these lost habitats. BC strategies involve a suite of actions to conserve and restore BC habitats to contribute to climate mitigation (Duarte et al., 2013). BC habitats are efficient at long-term sequestration of organic carbon by burying a fraction of their own production in the soils/seabeds (Krause-Jensen and Duarte, 2016). By altering turbulence, flow and wave action, these habitats promote sedimentation and accumulate significant quantities of allochthonous carbon and nutrients (Kennedy et al., 2010). While BC strategies are rooted in angiosperm-dominated coastal habitats, there is increasing recognition of the role that seaweeds, both wild and cultivated, can play in climate change mitigation and adaptation (Duarte et al., 2017; Froehlich et al., 2019; Krause-Jensen and Duarte, 2016; Krause-Jensen et al., 2018; Lovelock and Duarte, 2019) and the need to increase our understanding on the role of seaweed in carbon sequestration and the fate of their carbon (Macreadie et al., 2019). Seaweeds release both particulate and dissolved organic carbon (Hill et al., 2015; Krause-Jensen and Duarte, 2016), which can be buried into sediments or transported into deep sea, thus acting as a CO_2 sink (Duarte et al., 2017; Ortega et al., 2019; Queirós et al., 2019). In contrast to angiosperm-dominated coastal habitats, which are declining globally (Duarte et al., 2013), wild seaweed communities seem to be globally stable, with declines in some areas being compensated by expansions elsewhere (Krumhansl et al., 2016). Meanwhile, the global growth of seaweed aquaculture, now covering 1600 km^2 globally (Duarte et al., 2017), provides food and raw materials to large segments of the human populations, and offers an emerging opportunity to contribute to BC-based strategies (Duarte et al., 2017; Krause-Jensen et al., 2018; Lovelock and Duarte, 2019) because of its scalability in contrast to the limited scope for angiosperm-dominated BC strategies (Gattuso et al., 2018).

China has an extensive coastline, spanning 18,000 km along its continental area and 32,000 km when including its islands. The coastline has experienced abrupt transformation, including massive coastal habitat loss involving about 51% of coastal wetlands and 69% of mangrove forests loss due to reclamation, mostly over the past 20 years (Zhang et al., 2005). On the other hand, China accounts for about 70% of global seaweed aquaculture (FAO, 2010). Hence, China meets the criteria to adopt BC strategies within the range of policies best suited to respond to climate change. China is now preparing to join the growing pool of nations that have adopted and developed national BC programs. Indeed China has already included BC actions among its Nationally Determined Contributions (NDCs) (Gallo et al., 2017; Herr and Landis, 2016), which represent the basic building blocks of national strategies for implementing the Paris Agreement and reflect the highest possible ambition of the nations to mitigate climate change (Gallo et al., 2017). A recent assessment reported that 27 nations, including China, have included Blue Carbon mitigation contributions in their NDCs, encompassing ocean carbon storage and the protection, replantation, or management of mangroves, saltmarshes, and seagrass (Gallo et al., 2017). For example, "Mangrove in South and *Tamarix chinensis* in North" project which was started in 2016 by the State Oceanic Administration is mainly focused on the planting mangrove trees in south China and planting saltmarshes vegetation, such as *Tamarix chinensis, Phragmites australis,* and *Suaeda salsa* in north China (Ministry of Natural Resources, 2016). However, there is ample scope to broaden the slate of Blue Carbon actions included in NDC's (Gallo et al., 2017). In this context, wild and farmed seaweed are not yet included in NDC's, as further research to document their contribution to carbon sequestration is required before emissions reduction factors can be used in supporting the potential NDCs involving seaweed management (Krause-Jensen et al., 2018; Froehlich et al., 2019; Lovelock and Duarte, 2019).

Here we identify opportunities for BC approaches, including vegetated coastal habitats as recognized BC habitats and also the emerging role of seaweed aquaculture as potential BC habitats, to help China mitigate and adapt to climate change, thereby developing a roadmap that makes use of the unique characteristics and capacities of China. We first characterize the wild and created BC habitats in China, and examine the changes occurring since 1950, and the drivers for these changes. We then consider the specific case for the expansion of seaweed aquaculture as a unique emerging BC resource in China (Duarte et al., 2017), and the requirement for engineering solutions to enhance its potential. Lastly, we consider the scope for conservation and restoration of BC habitats in the light of its consistency with national policies, and identify the climate-change mitigation and adaptation potential as well as environmental and economic co-benefits of the development of BC strategies in China.

2. Wild and created blue carbon habitats in China

Along the 32,000 km of Chinese coastline, BC habitats (i.e. tidal saltmarsh, mangrove and seagrass ecosystems) together with seaweed farms extend over 3000 km^2, about 60% of which are occupied by wild habitats (i.e. habitats that exist without artificial cultivation) and about 40% are created habitats (i.e. habitats that have been artificially created and maintained). Wild tidal saltmarshes occupy about 1500 km^2 (about 82% of wild BC habitats) (Guan, 2012; Zhang et al., 2005; Zuo et al., 2013), while mangroves and seagrasses account for 13% (235 km^2) and 5% (88 km^2) of wild BC habitat extent, respectively (Zheng et al., 2013). The area of wild seaweeds in China remains unknown (Table 1), while seaweed farms extend across 1250 km^2 of China's coast (Xiao et al., 2017) and represent more than 99% of its created BC habitats and around 41% of the extension of all (wild and created) current BC habitats (Table 1), with the extent of seaweed farms growing rapidly.

While saltmarshes are distributed all along the entire coast of China, mangroves are restricted to the southeastern China and seagrasses occur in both the North and South (Fig. 1). Eight tidal saltmarsh species have been reported in China (Table 1). *Phragmites australis* and *Suaeda salsa* dominate the native tidal saltmarsh vegetation and the invasive species *Spartina alterniflora* accounts for about 28% of the total tidal saltmarsh extent (Zhang and Shi, 2007; Zuo et al., 2012). Mangrove forests in China have a high biodiversity with 20 different species present, while five seagrass species have been reported (Table 1).

Organic carbon (C_{org}) stocks in wild mangrove, tidal saltmarsh and seagrass ecosystems in China are estimated to be 12–36 Tg C_{org}. Mangrove forests store 39–60% of Chinese BC while tidal saltmarshes and seagrasses store 28–35% and 12–25%, respectively. The total C_{org} sequestration rates in wild mangrove, tidal saltmarsh and seagrass ecosystems are estimated to be 0.32–0.64 Tg C_{org} yr^{-1}. The areal carbon sequestration by mangroves has been estimated at about 4.44 Mg C_{org} ha^{-1} yr^{-1}, about 2.36 Mg C_{org} ha^{-1} yr^{-1} for tidal saltmarsh, and 0.024–0.101 Mg C_{org} ha^{-1} yr^{-1} for seagrass ecosystems (Table 1). Carbon storage of mangrove is much more than the carbon stored in saltmarsh and seagrass sediments.

Since 1950, the extent of mangrove forests and tidal saltmarshes in China has declined rapidly. The extent of mangroves between 1950 and 2002 has declined at an overall national rate between 1.2% yr^{-1} (Wilkie and Fortuna, 2003) and 5.1% yr^{-1} (Fu et al., 2009), averaging 3.5% yr^{-1} and, overall, 77–87% of these habitats have been lost since 1950 (Table 1). The extent of 8 saltmarshes distributed along the entire Chinese coast between 1980 and 2013 decreased at rates varying between 12.2% yr^{-1} and 37.8% yr^{-1}, but when considering the initial size of individual tidal saltmarshes prior to onset of coastal development, they overall declined at an average weighed rate of 3.2% yr^{-1} (Table 1). There are no records of temporal changes in the extent of seagrass habitat along Chinese coast. Niu et al. (2012) estimated that 65.4% of the coastal swamps have been lost between 1978 and 2008, suggesting that extensive areas of seagrass have also been lost. Overall, China experienced a massive development over the last half century, which resulted in the loss and degradation of BC ecosystems at rates of 3.2–3.5% yr^{-1} since 1950 (Table 1). Tidal saltmarsh, mangrove and presumably seagrass losses have been related to coastal development activities, including agriculture, aquaculture, and land reclamation for aquaculture ponds, paddy fields, industry and urban construction (Zhang et al., 2005).

Although the areal extent of created BC habitats in China does not compensate for the losses since 1950, it already accounts for 14% of the estimated losses. This is mainly due to seaweed farming, which has grown at a rate of about 8% yr^{-1} since 1950 (Xiao et al., 2017). Since the 1990s, the Chinese government has successfully restored, by planting, 15 km^2 of mangrove forests (Chen et al., 2009). There remains a great potential for mangrove expansion along Chinese coasts. Since 2006, the State Forestry Administration and State Oceanic Administration have identified 656 km^2 of intertidal zones suitable for mangrove afforestation (Chen et al., 2009). Seagrass planting in small-scale trials was conducted between 1989 and 2008, but it was restricted to artificial reefs (Shu et al., 2011) and aquaculture ponds to improve environmental conditions for shrimp and sea cucumber cultures (Ren et al., 1991). However, seagrass planting failed as majority of seagrass was lost in 1–5 years after planting (van Katwijk et al., 2016). The current capacity of C sequestration by wild BC ecosystems in China is estimated to be 65–80% lower than before 1950. In this respect, the conversion of mangrove and saltmarshes to aquaculture ponds may have emitted 15–82 Tg CO_2 eq yr^{-1}.

Different from the great loss of vegetated coastal habitats, seaweed farming in China increased from 1978 (269,000 tons of dry weight) to present (1,885,000 tons of dry weight in 2014) (Xiao et al., 2017). Farmed seaweeds capture into biomass 3.97 Mg C_{org} ha^{-1} yr^{-1}, but the area of wild seaweed in China remains unknown and therefore, it was not possible to estimate their C_{org} sequestration capacity at the national scale. Assuming that 24.8% of the seaweed biomass is C_{org} (Duarte, 1992), farming seaweed in China could result in the mitigation of 0.5 Tg C_{org} yr^{-1} (1.82 Tg CO_2 yr^{-1}), assuming that all C_{org} stored in seaweed biomass be preserved or converted into biofuels, which represents an upper limit that is currently far from being met as much of the biomass is allocated to human consumption. The potential C mitigation capacity of seaweed (3.97 Mg C_{org} ha^{-1} yr^{-1}) is comparable to the sequestration rate of other BC ecosystems. The current CO_2 removed by farming seaweed is equivalent to 0.01–0.03% of fossil fuel-CO_2 emissions in China (Fig. 2). However, only a small fraction of the CO_2 removed by farmed seaweed may be possibly stored at present, although there is ample potential to increase this contribution through the development of biofuels, biochar and soil-amendment industries based on seaweed (Duarte et al., 2017; Froehlich et al., 2019). Moreover, seaweed aquaculture leads to avoided emissions as the CO_2 footprint of seaweed aquaculture is much lower than that of producing equivalent amounts of food on land (Duarte et al., 2017; Froehlich et al., 2019; Zheng et al., 2019).

3. Seaweed aquaculture as a blue carbon resource in China

Seaweeds are the dominant primary producers in the coastal zone which play an important role in CO_2 removal (Duarte and Cebrián, 1996), producing an estimated 1.5 Pg C $year^{-1}$ of dry products over the 3.4 million km^2 they cover (Krause-Jensen and Duarte, 2016). Despite their major contribution to ocean production, seaweeds have generally not been considered to contribute to marine carbon sinks as they typically grow on rocky substrates that do not accumulate C_{org}. However, seaweeds release considerable amounts of detritus, both as particulate (Colombini and Chelazzi, 2003; Duarte and Cebrián, 1996; Duggins et al., 2016; Filbee-Dexter and Scheibling, 2012; Krumhansl and Scheibling, 2012), and dissolved organic carbon (DOC) (Barron et al., 2014), comprising about 43.5% of seaweed production (Duarte and Cebrián, 1996). Seaweed C_{org} can subsequently be transported across large distances both as DOC (Barrón and Duarte, 2016), particularly if converted by microbes into recalcitrant DOC forms (Jiao et al., 2010), and particulate C_{org} (POC) (Krause-Jensen and Duarte, 2016) to reach ocean carbon sinks (Krause-Jensen and Duarte, 2016; Hill et al., 2015). Despite earlier assumptions that seaweed C_{org} was not sequestered into long-term reservoirs (Miller et al., 2009), recent assessments estimate potential seaweed contributions to long-term C_{org} sequestration in sediments and the deep sea at 173 Tg C yr^{-1} (Krause-Jensen and Duarte, 2016), which render seaweed carbon an emerging component of blue carbon strategies (Krause-Jensen et al., 2018; Lovelock and Duarte, 2019). Moreover, Ortega et al. (2019) reported that 25% of exported macroalgal carbon is sequestered in long-term reservoirs, such as coastal sediments and the deep sea. Hence, the potential seaweeds contribution to global C_{org} sequestration is comparable to that for saltmarshes, seagrass beds and mangroves combined. Therefore, the potential of seaweeds to support C_{org} sequestration could provide, when integrated with large-scale seaweed aquaculture appropriately managed to mitigate climate change, an option for climate mitigation (Duarte et al., 2017).

Although >1200 species of seaweeds have been described from Chinese coastal waters (Zeng, 1962; Zhang, 1996), their naturally occurring biomass is poorly constrained. The harvest of seaweed and

Table 1
Wild and created Blue Carbon habitats. Past and current extent, organic carbon (Corg) stocks in 1 m-thick deposits, C sequestration (seq.) rates, absolute habitat area change (Δarea), period of habitat change, relative rate of area change, causes of loss (wild habitats) and cause of recovery (re-created habitats), % of the lost wild habitats converted (conv.) to aquaculture ponds, and associated potential CO2 equivalents emissions (emis.) from wild habitat conversion to aquaculture ponds and potential CO2 equivalents seq. by created habitat in China. * Carbon sequestration represents the living biomass (not the soil); ** % habitat conversion to aquaculture ponds assumed to be the same as for tidal saltmarsh. nd: no data.

Habitats/Species	Current habitat extent (km^2)	Soil C stock (Mg C_{org} ha^{-1})	Soil C stock in China (Tg C_{org})	Soil C seq.(Mg C_{org} ha^{-1} yr^{-1})	Soil C seq. in China(Tg C_{org} yr^{-1})	Habitat extent before 1950 (km^2)	Δarea (km^2)	Period of change	Rate of change (% yr^{-1})	Cause of loss	Habitat conv. to aquaculture ponds (%)	CO_2 emis. by habitat conv. to aquaculture ponds/CO_2 seq. by created habitat in China (Tg CO_2 eq yr^{-1})
1. WILD HABITATS												
Mangrove (Fu et al., 2009)	31(Hamilton and Casey, 2016) to 344 (Peng et al., 2013)	186	0.58 to 6.40	4.44	0.014 to 0.153	2500 (Lu et al., 2016)	−2156 to-2469	Before 1950 to 2014	−3.5 (Fu et al., 2009; Brennan, 2011)	agriculture, aquaculture, industry and urban construction (Chen et al., 2009)	26.32** (Jia et al., 2015)	3.45 to 20.25 (Kauffman et al., 2014; Sidik and Lovelock, 2013)
Tidal saltmarsh	1207 (Xu et al., 2018) to 1717 (Duan and Fei, 2008)	88 to 167	10.67 to 28.75	2.36	0.28 to 0.40	8797 (Duan and Fei, 2008)	> −7080 to > −7590 (Xu et al., 2018)	Before 1950 to 2013	−3.2 (Jia et al., 2015; Sun et al., 2015; Zhang et al., 2015; Yu et al., 2012; Xu et al., 2018)	reclamation, aquaculture, paddy fields (Jia et al., 2015)	26.32	11.33 to 62.24 (Kauffman et al., 2014; Sidik and Lovelock, 2013)
Seagrass	88 (Duan and Fei, 2008)	38 to 120 (Miyajima et al., 2015)	0.33 to 1.06	0.024 to 0.101 (Miyajima et al., 2015)	0.00021 to 0.00089	Nd	nd	nd	nd	nd	Nd	nd
Seaweed*	nd	Nd	Nd	nd	nd	Nd	nd	nd	nd	nd	Nd	nd
TOTAL wild-min	1326		**11.58**		>**0.29**	>**11,297**	>**-9236**					>**14.78**
TOTAL wild-max	**2149**		**36.21**		>**0.55**		>**-10059**					>**82.49**
2. CREATED HABITATS										Cause of recovery		
Mangrove	>2 (Fu et al., 2009) to 15 (Chen et al., 2009)	186	0.04 to 0.28	4.44	0.001 to 0.007	0	>2 to 15	nd		planting		0.006 to 0.15
Tidal saltmarsh	nd	Nd	nd	nd	nd	Nd	nd	nd	nd			nd
Seagrass (Ren et al., 1991; Shu et al., 2011)	nd	Nd	nd	nd	nd	Nd	nd	nd	nd			nd
Seaweed	1250 (Xiao et al., 2017)	Nd	nd	3.97 (Xiao et al., 2017; Duarte, 1992)	0.5 (Xiao et al., 2017; Duarte, 1992)	<2 (Xiao et al., 2017)	1248	1955 to 2013	8	farming		1.82 (Xiao et al., 2017; Duarte, 1992)
TOTAL created-min	**1252**					**2**	**1250**					
TOTAL created-max	**1265**					**2**	**1263**					
TOTAL min	>**2504**					>**11,299**	>**-7973**					
TOTAL max	>**3414**						>**-8809**					

Wild species: Mangroves: *Bruguiera gymnorrhiza, B. sexangula, Ceriops tagal, Kandelia obovata, Rhizophora apiculata, R. Stylosa, Acrostichum aureum, A. Speciosu, Acanthus ebracteatus, A. Ilicifolius, L. littorela, L. racemosa, Excoecaria agallaocha Xylocarpus granatum, Aegiceras corniculatum, Nypa frticans, Scypiphora hydrophyllacea, Sonneriatia alba, S caseolaris, S. hainanensis, S. ovata, S. gulngai, Heritiera littoralis Avicennia marina;* Saltmarsh: *Phragmites australis, Spartina alterniflora, Suaeda salae, Scripus mariqueter, Suaeda heteropter, Tamarix chinensis, Aeluropus sinenis, Imperata cylindrical;* Seagrasses: *Zostera marina, Thalassia hemprichii, Enhalus acoroides, Halophila ovalis, Phyllospadix iwatensis.*
Planted species: Mangroves: *B. sexangula, Kandelia obovata, R. stylosa, L. racemosa, S caseolaris, S. apetala, Heritiera littoralis;* Seagrasses: *Zostera marina, Zostera noltii, Halophila ovalis, Rupia maritima;* Seaweeds: *Laminaria, Undaria, Porphyra, Gracilaria, Eucheuma, Sargassum fusiforme, Ulva;* aquaculture ponds refer to fish/shrimp/crap/shell fish farming.

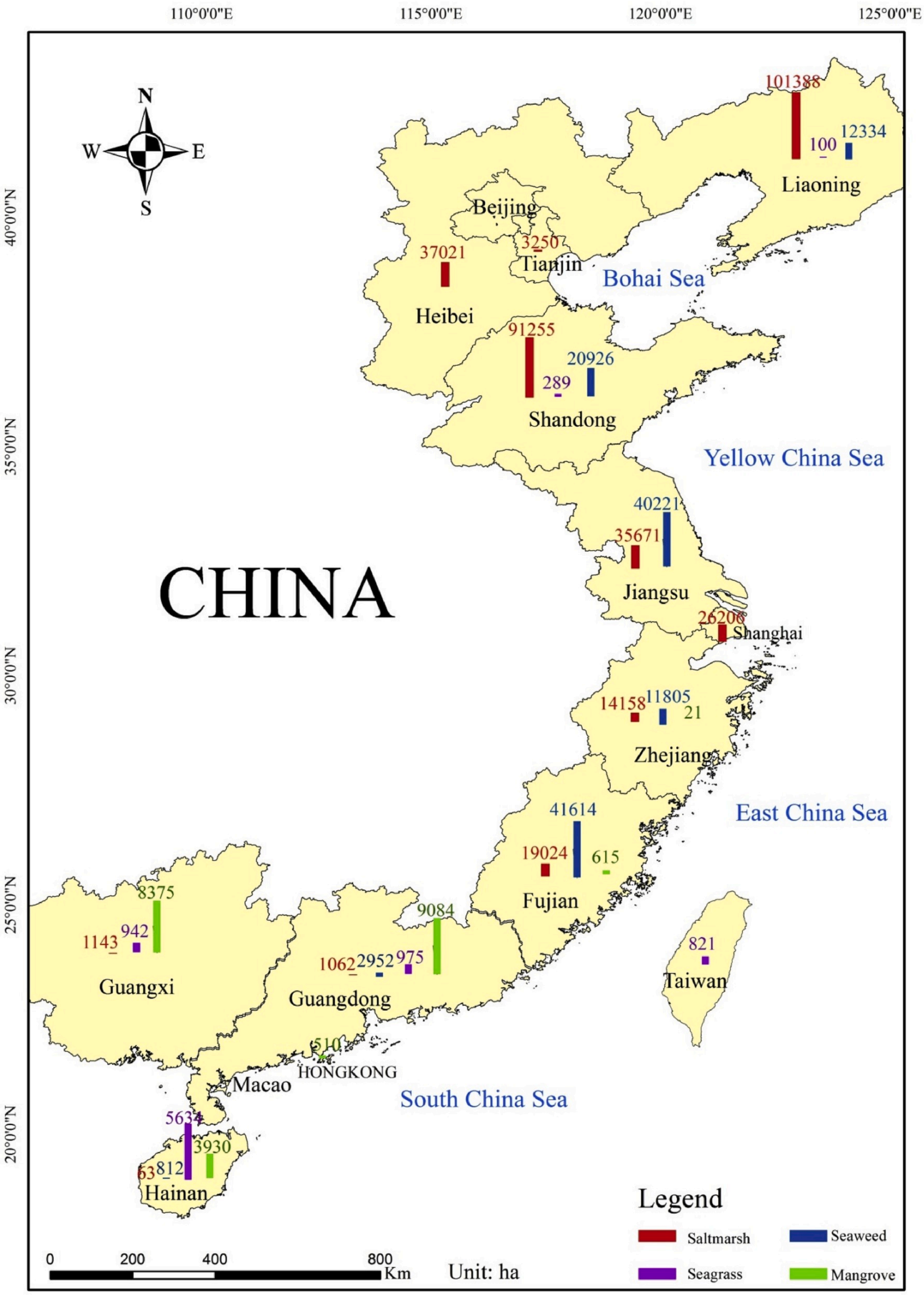

Fig. 1. Distribution of the area of Blue Carbon habitat in coastal provinces, China (figures in hectares). Data from Gu et al. (2018) for saltmarsh in 2015; China Fishery Statistical Yearbook 2016 (BFMA, 2016) for seaweed in 2015; Zheng et al. (2013) for seagrass from 1990 to 2010; Liao and Zhang (2014) for mangrove in 2001.

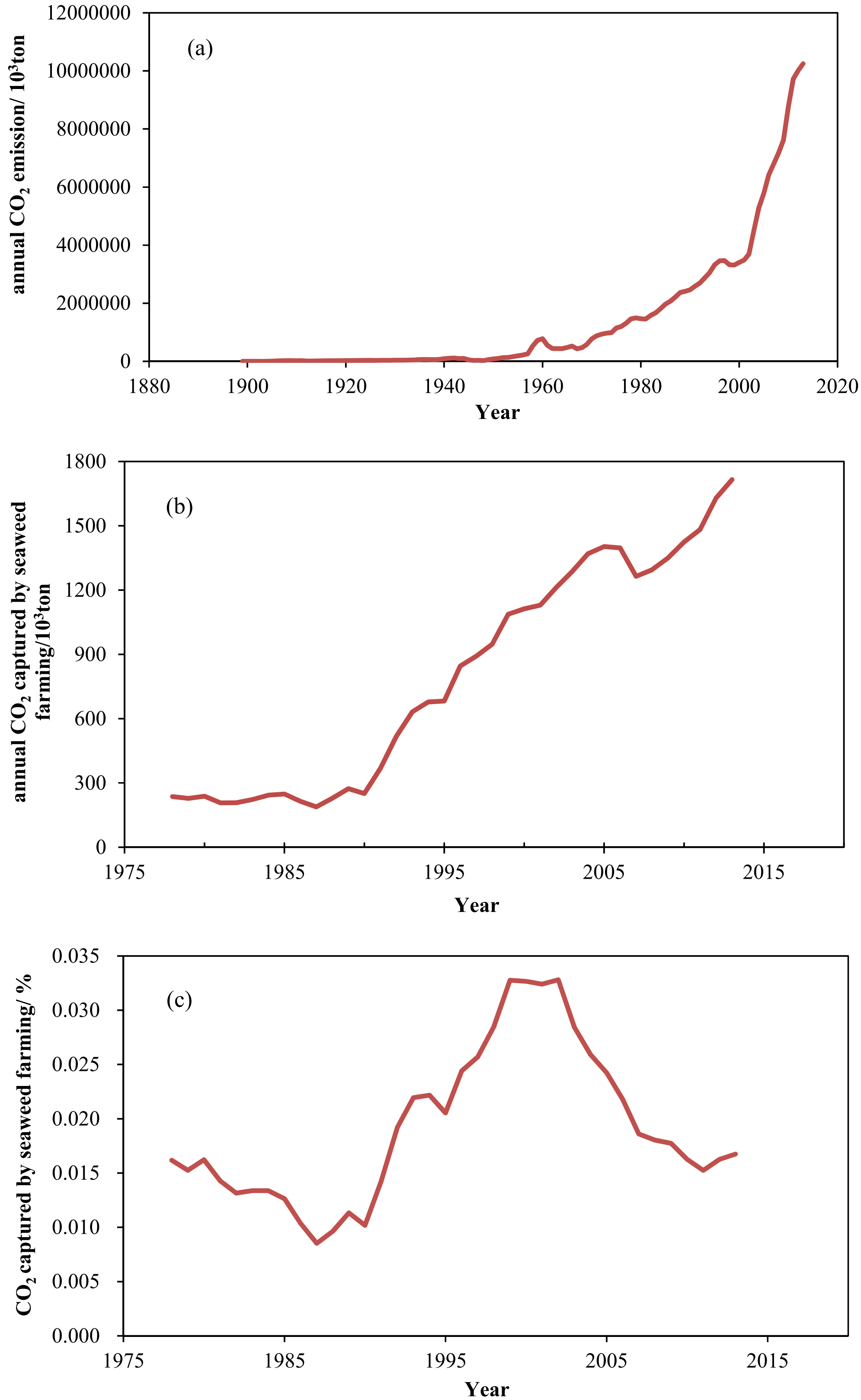

Fig. 2. (a) Annual Fossil Fuel CO_2 emissions in China; (b) annual CO_2 captured by seaweed farming in China; (c) the percentage of CO_2 captured by seaweed farming in China in relation to Fossil Fuel CO_2 emissions in China.

other aquatic plants in 2014 exceeded 13.1×10^6 t wet weight (Ye et al., 2017). Over 98% of this harvest was from seaweed aquaculture, with <2% being harvested from natural habitats (BFMA, 2015). Seaweeds have been widely used as sources of human and animal foods, fertilizers, pharmaceuticals, nutraceuticals, and biofuels and as biofilters to remove nutrients from coastal waters (Chopin et al., 2010; Fang et al., 2016; He et al., 2008; Neori, 2008; Troell et al., 1999; Xiao et al., 2017). Seaweed farming provides food with low CO_2 footprints and serves as fertilizer with much reduced CO_2 footprint relative to synthetic fertilizer production (Duarte et al., 2017; Xiao et al., 2017; Zheng et al., 2019). In 2015, the macroalgae farming area in China reached 1250 km^2, and the total yield was 2 million tons in dry weight, which removed about 75, 371 tons N and 9496 tons P from coastal waters (Xiao et al., 2017). Seaweed farming also saved an estimated nearly 1180 million tons freshwater for irrigation, and 171,958 tons N, 102,120 tons P and 133, 217 tons for chemical fertilizer in 2015 relative to land-based farming of a similarly large biomass, equivalent to about 1848 million Yuan RMB.

In other jurisdictions, macroalgae have been recognized as a viable option for C_{org} capture and storage (Chung et al., 2013; Sondak and Chung, 2015), this has, so far, not been the case in China. For example, due to seaweed cultivation as a significant CO_2 sink, Korea has developed Coastal CO_2 Removal Belts (CCRB), both natural and man-made plant communities in coastal regions of South Korea, to enhance CO_2 removal by seaweed forests (Chung et al., 2013). When populated with the perennial brown alga *Ecklonia*, a pilot CCRB farm can draw down 10 t of CO_2 ha^{-1} yr^{-1}, some of which can be potentially sequestered (Duarte et al., 2017).

The high efficiency of nutrient removal by seaweed aquaculture has been projected to possibly result, at current growth rates, in nutrient depletion and limitation beyond a doubling of the current area (Xiao et al., 2017). Hence, provided current nutrient inputs, the maximum carrying capacity of Chinese coastal waters to support seaweed aquaculture will be reached in less than a decade (Xiao et al., 2017). Currently, seaweed aquaculture is deployed in coastal areas where the farms intercept nutrient inputs from land and, therefore, alleviate eutrophication, typically reducing nutrient concentrations by about half (Xiao et al., 2017). Added benefits of reducing eutrophication involve mitigation of hypoxia, which results from eutrophication, through both removing nutrients and directly injecting oxygen in coastal waters from seaweed photosynthesis. CO_2 removal by seaweed farms also contributes to raise pH and alleviate ocean acidification (Chung et al., 2013; Duarte et al., 2017; Hendriks et al., 2014).

The total area of enclosed or semi-enclosed bays in China is about 27,760 km^2, with about 5830 km^2 (i.e. 20%) currently in use for mariculture, of which seaweed aquaculture comprises 20% of the total area (BFMA, 2016). However, 90% of the area in bays suitable for aquaculture is currently occupied, thus the scope for expansion is limited (Zhang et al., 2012a,b). Major expansion of seaweed aquaculture is possible either within the geographic footprint by polycultures with animal aquaculture, or the geographic expansion into exposed bays or offshore waters which are unsuitable under current practices. Below we propose a range of technological approaches that will increase the capacity of China to enhance seaweed production and, therefore, its potential role in climate change mitigation.

Further expansion of seaweed aquaculture to increase their potential for climate change mitigation would require seaweed aquaculture to expand offshore where nutrients can be supplied through technologies such as artificial upwelling (AU) powered by renewal energy and avoiding the risks of eutrophication and hypoxia in more nearshore coastal waters. AU transports cold, nutrient- and CO_2-rich waters from below the thermocline to the euphotic zone where the nutrients and CO_2 can be assimilated by the seaweeds with concomitant drawdown of CO_2, and the production of plant biomass (Pan et al., 2015; Fan et al., 2016). The technique has been implemented in Aoshan Bay, Qingdao, China, a semi-closed bay (Table 2). The surface seawater is oligotrophic, and the sediments contain high nitrogen and phosphorus. Cultivation of *Laminaria* leads to nutrient limitation in spring due to the strong absorbing ability of seaweed, which can be overcome through nutrients delivered through artificial upwelling systems. This technique has also been applied in western Norwegian fjords to pump deep water to the surface to enhance nutrient concentrations and stimulate phytoplankton growth in an attempt to enhance fisheries production (Aure et al., 2007; Handå et al., 2013; McClimans et al., 2010).

In addition to nutrient availability, further expansion of seaweed aquaculture in China, and, thus, its emerging contribution to climate change mitigation is limited by current farming practices requiring sheltered conditions. High exposure to waves and strong tidal currents render many areas in coastal China seas unsuitable to support seaweed aquaculture (Burrows, 2012; Norderhaug et al., 2014; Tuya and Haroun, 2006). For example, during 2010 there were over 130 storm surges in Chinese coastal waters (Zhang et al., 2012a), which could have potentially damaged seaweed farms if they were in exposed locations. However, wave-absorbing devices can dissipate this energy, thereby allowing seaweed farming while also supplying mechanical energy to power AU that delivers nutrients to the surface layer or circulate the biomass to enhance exposure to light and, therefore, maximize yield. The wave energy technology has been successfully tested in Dongtou county, where *Sargassum fusiforme* farms are distributed (Table 2).

Offshore seaweed aquaculture in high energy environments can also be supported by implementing efficient anchoring systems together with buoyancy regulation systems to lower aquaculture rafts to depths protected from excessive wave action during storms, and raise the rafts subsequently. The anchoring system is essential in *Sargassum fusiforme* farming, which is always used to fix the rafts. This has been in practice in Dongtou county (Table 2). Prototypes of self-contained buoyancy regulation systems have been tested in *Sargassum fusiforme* farm. The buoyancy regulation systems are also being constructed to support offshore seaweed mariculture in New Zealand, in USA and in Germany (Goseberg et al., 2017). Hence, there is no technical barrier preventing buoyancy regulation systems to be used for macroalgae farming. The

Table 2
Summary of engineering solutions and innovative technologies to support the expansion of seaweed aquaculture in China.

Engineering solutions/ innovative technologies	Challenges in expanding seaweed aquaculture	Functions	Increasing	Implemented/Tested locations in China
Artificial upwelling (Maruyama et al., 2004)	Nutrient limitations due to strong absorbing ability of seaweed.	Bring high nutrient deep seawater to surface layer where seaweed grows.	Area, Yield	*Laminaria* cultivation in Aoshan Bay, Qingdao City
Anchoring system (Roesijadi et al., 2008)	Extension of seaweed farm from coastal to offshore.	Offer platform for offshore seaweed aquaculture.	Area	*Sargassum fusiforme* cultivation in Dongtou county
Artificial light supplementary	Light limitations in Chinese coast due to high water turbidity, or long and continuous cloudy/rainy period of weather.	Promote seaweed growth and biosynthesize of targeted bio-molecular, increasing the value of seaweed products.	Yield	
Turn-over aquaculture device	Lack of habitat for intertidal seaweed species.	Providing artificial dry exposure condition for seaweed	Area	*Porphyra* cultivation in Dayu Bay, Cang-nan County
Wave energy technology	Too strong waves in seaweed aquaculture area.	Dissipate turbulence energy via wave absorbing.	Area	*Sargassum fusiforme* cultivation in Dongtou county
Buoyancy regulation system	Damage due to storms.	Mechanically lowered and raise aquaculture rafts to adjust the depth.	Area	*Sargassum fusiforme* cultivation in Dongtou county

remaining issue is to lower the cost as to make the seaweed industry profitable. If successful, this development will allow large scale, sustainable seaweed farming, which, if properly managed, can contribute to climate change mitigation. However, these technologies would add costs to seaweed aquaculture, which may not be viable under the current market-based cost model. However, accounting for the greenhouse mitigation services of seaweed aquaculture through carbon credits, for which farmers are currently not compensated, may provide the additional income to afford the costs of deploying these engineering solutions. Hence, realizing the potential of seaweed aquaculture to contribute to climate change mitigation requires market and policy interventions and not only engineering solutions.

One good example for technological development already in place is the turn-over aquaculture device for *Porphyra* cultivation, developed in 2010 to provide artificial dry exposure conditions for *Porphyra*, which, in turn, enables greatly the extension of *Porphyra* farms from inter-tidal zone to near-coast, and to offshore (Table 2). This device has been implemented for years in Cangnan county, Zhejiang province, and in Fuding county, Fujian province, supporting the expansion of *Porphyra* cultivation, which provides high profit but is currently limited by the lack of habitat.

Most seaweed aquaculture yield in China is currently allocated to human food supply. This only marginally contributes to climate change mitigation through avoiding emissions associated with the production of similar food amounts in land-based agriculture which has a larger greenhouse gas footprint (Duarte et al., 2017). However, maximizing climate change mitigation through seaweed aquaculture requires that seaweed yield would be used for e.g. biofuel production (Duarte et al., 2017), long-lasting products and use of remaining waste for biochar production for soil amelioration (Bird et al., 2011). Yet, an industry for biofuel production from seaweed aquaculture, or long-lived seaweed based products, is currently lacking in China (Wei et al., 2013).

4. Rationale and elements for a blue carbon program in China

The government of China is committed to slow down or even reducing CO_2 level as a commitment in the Paris Agreement, and to establish healthier ecosystems, for which it has invested tremendous efforts in marine ecosystem restoration. The 12th Five-Year Plan of National Marine Development (2013) and the "Mangrove in South and *Tamarix chinensis* in North" project in the 13th Five-Year Plan (2016–2020) illustrate this commitment at national level. The Fifth Plenary Session of the Eighteenth Central Committee of the Communist Party of China (26–29 October 2015) approved the "Blue Bay Project", setting goals for the restoration of coastal habitats (Ministry of Natural Resources, 2016). All these projects listed restoration of coastal vegetation as a national priority. Secretary General Xi reports to the Nineteenth Congress of the Communist Party of China (18–24 October 2017) included a chapter dedicated to the ocean, calling for (1) integrated land-ocean management; (2) enhanced efforts to address key marine environmental issues, protect shorelines and prevent coastal disasters; and (3) strengthen protection and restoration of coastal wetlands by joining global environmental initiatives. A national BC strategy would align with this aim while expanding the scope of existing national strategies (Table 3).

Many nations, both developed and developing ones, have defined national BC programs (e.g. Australia, France, Japan, Indonesia, Malaysia, Saudi Arabia), and China is now developing its national BC program. A BC strategy for China meets the criteria of being readily actionable, cost-effective and generating co-benefits, in terms of adding value towards existing national priorities (Table 3). We identify the following reasons supporting a national BC program in China:

1. China has lost about 77–87% (Table 1) of the natural BC habitat, with great impacts on biodiversity, ecosystem health and environmental quality.

Table 3
China's national policies aligned with Blue Carbon strategies.

Projects	Goals	Status quo
The 12th Five-Year Plan of National Marine Development	200 km^2 new wetland (100 km^2 mangroves, and 100 km^2 *Phragmites australis* wetland).	The area of mangrove in China decrease from about 420 km^2 in 1950s to 345 km^2 in 2013.
"Mangrove in South and *Tamarix chinensis* in North"Project	2500 ha mangrove in south China, 4000 ha *Phragmites australis*, 1500 ha of *Suaeda salsa*, and 500 ha of *Tamarix chinensis* in north China	
Marine Ecological Redline	Natural coastline should be no less than 35% and coastal waters of good water quality (case one or case two) should reach the proportion of about 70% by 2020.	The China natural coastline keeps declining since 1940s, and there is less than 30% left in 2014; the case one and case two waters in coastal area are 33.6% and 36.9% respectively.
"Blue Bay" renovation project	Enlarge the area of coastal wetland and meet environmental standard in bay areas.	Between 2000 and 2010, the area of coastal wetland decreased by 3288 km^2, and the artificial wetland increased by 2592 km^2.

2. China is already investing heavily in the restoration and conservation of BC habitats, such as mangrove and saltmarsh habitats (Table 3), but is only recently considering computing the carbon mitigation value associated with these projects in its Nationally Determined Contributions. Accounting for this on-going carbon sequestration will help meet the commitments of China under the Paris Agreement.
3. China has developed a massive seaweed aquaculture industry, which has created thus far 1250 Km^2 of seaweed habitat, growing at 8% per year, with important – but yet unrealized - potential for climate change mitigation and adaptation.
4. The development of a BC program around seaweed aquaculture will catalyze the further growth of this blooming industry, which is delivering major benefits to Chinese economy and helping alleviate coastal eutrophication - a major national problem.
5. A national BC program will provide a cost-effective contribution to meeting China's objectives under the Paris Agreement.
6. A BC program may develop pioneer technology for carbon capture that can be exported elsewhere, generating additional value and opportunities for economic development.
7. Restoration and creation of coastal habitats will contribute to protect the vulnerable low-lying shorelines of China from sea level rise and storm surges, thereby avoiding losses of hundreds of lives and billions of RMB every year.
8. Restoration and creation of coastal habitats will contribute to generate nursery habitats for fish and other marine organisms of commercial value that will contribute to enhance the stocks and recover them from overexploitation.

Further, we propose that a BC program in China should consider the following elements:

1. Capacity building: Develop capacity within China's scientific community, both graduate students and early career researchers (e.g. junior faculty) and coastal management and policy agencies, to provide the knowledge, technology and policy frameworks supporting a national BC program.
2. Evaluation of BC resources: Evaluate the current extent, losses and gains of BC habitats, and the green-house gas emissions associated with these changes.
3. Demonstration of the value of seaweed aquaculture as a BC resource in China: Examine the CO_2 sequestration capacity of seaweed farms and the management and marketing options supporting a BC role.

4. Assessment of the contribution of restoration and conservation of BC habitats in China to national climate change policies.
5. Development of novel BC technologies: e.g. technologies to increase carbon capture by seaweed farms, and the potential use of marine plant litter to minimize Green House Gas emissions from agriculture.
6. Policy and Management: Development of policies and management tools to govern BC resources as to deliver the full potential of environmental benefits, involving fishermen and seaweed farmers in meeting the strategic objectives.
7. Establishment and improvement of nation-wide Carbon trade market: Based on the experience of existed carbon trade pilots, gradually and steadily establish nation-wide carbon trade market while including seaweed aquaculture into the carbon trade.

China's extensive coastline, loss of coastal habitats and vulnerability to climate change provide the opportunity, and necessity, to develop a strong BC program. While its contribution to address the nation's commitments under the Paris Agreement will be modest, the emerging BC program in China is poised to catalyze the restoration and conservation of coastal habitats, generating major benefits for all. Accordingly, the State Ocean Administration of China has taken the lead in BC actions in China through a series of actions, including compilation of a report on BC in China, released at the 2017 International Blue Carbon Forum held in Xiamen, China, November 4–5, 2017, the inclusion of BC in China's first biennial update report on climate change, preparation of a number of demonstration projects of technical standards for BC monitoring, and international cooperation in BC research with Thailand, Malaysia and Indonesia. With its extensive coastline and commitment, China is poised to play a key role in the implementation of BC strategies for climate change mitigation and adaptation.

Declaration of competing interest

Compliance with ethics guidelines: All authors (Jiaping Wu, Haibo Zhang, Yiwen Pan, Dorte Krause-Jensen, Zhiguo He, Wei Fan, Xi Xiao, Ikkyo Chung, Nuria Marbà, Oscar Serrano, Richard B. Rivkin, Yuhan Zheng, Jiali Gu, Xiujuan Zhang, Zhaohui Zhang, Peng Zhao, Wanfei Qiu, Guangcheng Chen, Carlos M. Duarte) declare that they have no conflict of interest or financial conflicts to disclose.

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