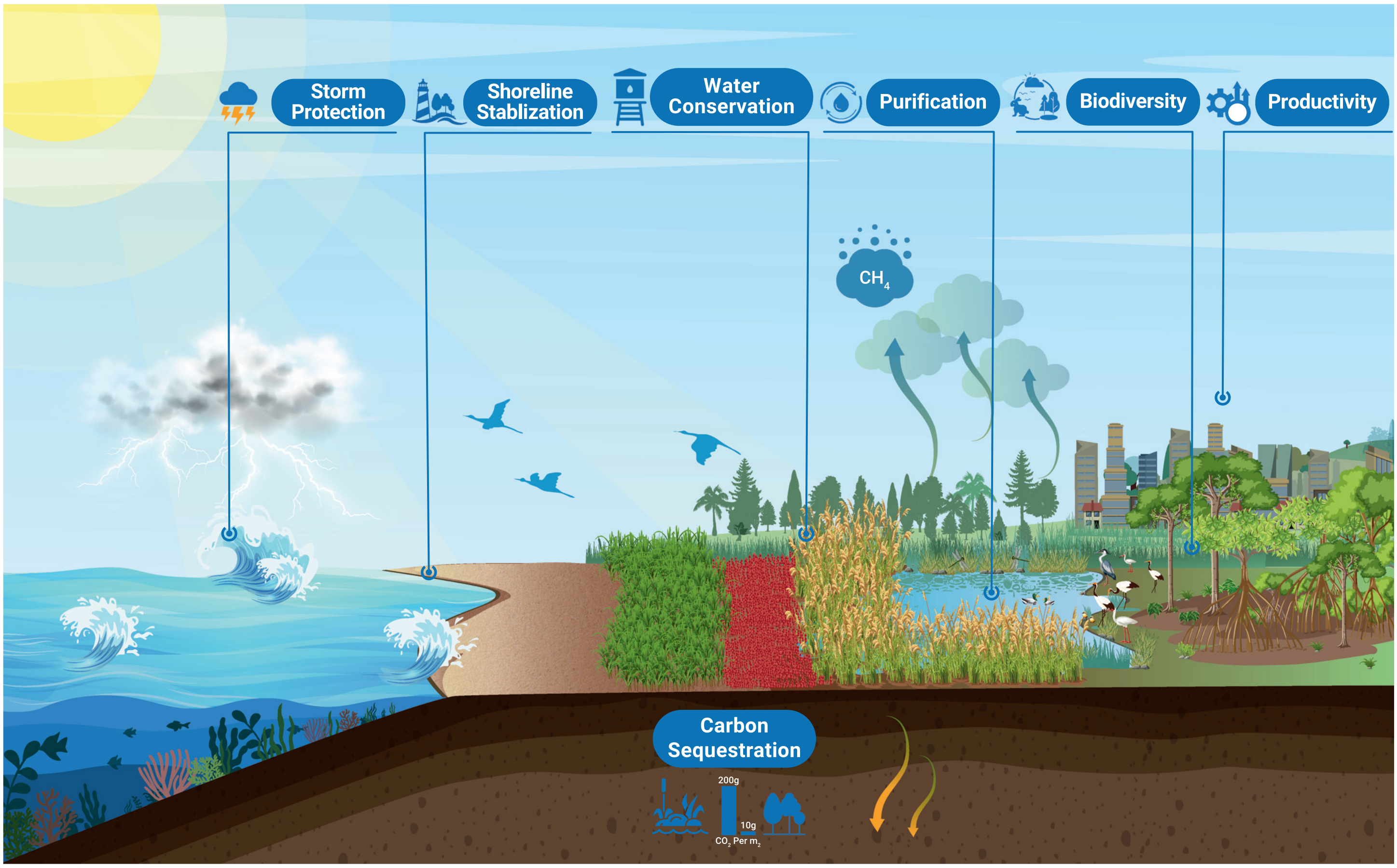
Coastal blue carbon in China as a nature-based solution toward carbon neutrality

Faming Wang,[1,2] Jihua Liu,[3] Guoming Qin,[1,2,4] Jingfan Zhang,[1,2,4] Jinge Zhou,[1,2,4] Jingtao Wu,[1,2] Lulu Zhang,[1,2] Poonam Thapa,[1,2] Christian J. Sanders,[5] Isaac R. Santos,[6] Xiuzhen Li,[7] Guanghui Lin,[8,9] Qihao Weng,[10] Jianwu Tang,[7] Nianzhi Jiao,[11,*] and Hai Ren[1,2,*]

*Correspondence: jiao@xmu.edu.cn (N.J.); renhai@scbg.ac.cn (H.R.)

Received: February 25, 2023; Accepted: July 9, 2023; Published Online: July 21, 2023; https://doi.org/10.1016/j.xinn.2023.100481

GRAPHICAL ABSTRACT



PUBLIC SUMMARY

- Protecting and restoring coastal ecosystems such as mangroves, salt marshes, tidal flats, and seagrass meadows can be a key strategy for China to achieve its goal of carbon neutrality by 2060.
- Coastal ecosystems in China alone have stored large amounts of carbon, approximately 118 Tg C.
- In addition to helping fight climate change, protecting these ecosystems also provides other cost-effective benefits, including storm protection, shoreline stabilization, water conservation, purification, high biodiversity, and productivity.

Coastal blue carbon in China as a nature-based solution toward carbon neutrality

Faming Wang,[1,2] Jihua Liu,[3] Guoming Qin,[1,2,4] Jingfan Zhang,[1,2,4] Jinge Zhou,[1,2,4] Jingtao Wu,[1,2] Lulu Zhang,[1,2] Poonam Thapa,[1,2] Christian J. Sanders,[5] Isaac R. Santos,[6] Xiuzhen Li,[7] Guanghui Lin,[8,9] Qihao Weng,[10] Jianwu Tang,[7] Nianzhi Jiao,[11,*] and Hai Ren[1,2,*]

[1]Xiaoliang Research Station of Tropical Coastal Ecosystems, Key Laboratory of Vegetation Restoration and Management of Degraded Ecosystems, and the CAS Engineering Laboratory for Ecological Restoration of Island and Coastal Ecosystems, South China Botanical Garden, Chinese Academy of Sciences, Guangzhou 510650, China
[2]Southern Marine Science and Engineering Guangdong Laboratory (Guangzhou), Guangzhou 510650, China
[3]Marine Research Institute, Shandong University, Qingdao 266237, China
[4]University of Chinese Academy of Sciences, Beijing 100049, China
[5]National Marine Science Centre, Faculty of Science and Engineering, Southern Cross University, Coffs Harbour, NSW 2450, Australia
[6]Department of Marine Sciences, University of Gothenburg, 41319 Gothenburg, Sweden
[7]State Key Laboratory of Estuarine and Coastal Research and Institute of Eco-Chongming, East China Normal University, Shanghai 201100, China
[8]Key Laboratory for Earth System Modeling, Ministry of Education, Department of Earth System Science, Tsinghua University, Beijing 100084, China
[9]Laboratory of Stable Isotope and Gulf Ecology, Institute of Ocean Engineering, Tsinghua's Shenzhen International Graduate School, Shenzhen 518055, China
[10]Department of Land Surveying and Geo-Informatics, The Hong Kong Polytechnic University, Hongkong 999077, China
[11]Innovative Research Center for Carbon Neutralization, Global ONCE Program, Xiamen 361005, China
*Correspondence: jiao@xmu.edu.cn (N.J.); renhai@scbg.ac.cn (H.R.)

Received: February 25, 2023; Accepted: July 9, 2023; Published Online: July 21, 2023; https://doi.org/10.1016/j.xinn.2023.100481

Citation: Wang F., Liu J., Qin G., et al., (2023). Coastal blue carbon in China as a nature-based solution toward carbon neutrality. The Innovation **4(5)**, 100481.

To achieve the Paris Agreement, China pledged to become "Carbon Neutral" by the 2060s. In addition to massive decarbonization, this would require significant changes in ecosystems toward negative CO_2 emissions. The ability of coastal blue carbon ecosystems (BCEs), including mangrove, salt marsh, and seagrass meadows, to sequester large amounts of CO_2 makes their conservation and restoration an important "nature-based solution (NbS)" for climate adaptation and mitigation. In this review, we examine how BCEs in China can contribute to climate mitigation. On the national scale, the BCEs in China store up to 118 Tg C across a total area of 1,440,377 ha, including over 75% as unvegetated tidal flats. The annual sedimental C burial of these BCEs reaches up to 2.06 Tg C year^{-1}, of which most occurs in salt marshes and tidal flats. The lateral C flux of mangroves and salt marshes contributes to 1.17 Tg C year^{-1} along the Chinese coastline. Conservation and restoration of BCEs benefit climate change mitigation and provide other ecological services with a value of \$32,000 ha^{-1} year^{-1}. The potential practices and technologies that can be implemented in China to improve BCE C sequestration, including their constraints and feasibility, are also outlined. Future directions are suggested to improve blue carbon estimates on aerial extent, carbon stocks, sequestration, and mitigation potential. Restoring and preserving BCEs would be a cost-effective step to achieve Carbon Neutral by 2060 in China despite various barriers that should be removed.

INTRODUCTION

To achieve The Paris Agreement goal of 1.5°C warming by the end of the century, humanity must achieve net-zero carbon dioxide (CO_2) emissions by the 2050s. In the United Nations' 76th High-Level Debate on September 22, 2020, Chinese President Xi pledged: "China will strive to peak CO_2 emissions before 2030 and achieve carbon neutrality before 2060." This will require, in addition to massive and rapid decarbonization, very significant changes in current land-use trajectories to minimize and subsequently reverse CO_2 emissions. Without this dual approach, the total achieved mitigation will be insufficient to achieve the Paris Agreement goal and prevent further climate change.

Nature-based solutions (NbSs) can support decarbonization, through protection, restoration, and sustainable management of natural carbon sinks and reservoirs. These NbSs were firstly defined as "actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" by the International Union for Conservation of Nature at its 2016 World Conservation Congress. Nature-based climate solutions have the potential to offer greater cost-effectiveness and scalability compared with technological alternatives. While options such as direct air capture, geological sequestration, and biochar (BC) production are still in the early stages of deployment and encountering substantial economic, social, and environmental challenges, nature-based approaches hold promise for effectively addressing climate change.[1] By 2050, NbSs are expected to rise to 10–18 Pg CO_2e per year.[1] This is a significant proportion (10%–18%) of the total mitigation needed.

Current NbS strategies for climate mitigation exhibit a disproportionate emphasis on green carbon (C), notably through the promotion of forest ecosystems and tree planting. The lack of attention toward coastal and ocean-based carbon sequestration remains a critical oversight in current NbS methodologies. The "Blue C" concept was originally referred to as C captured by marine ecosystems covering both coastal and open ocean ecosystems.[2] However, the ongoing research and development of blue C in the past decade have predominantly involved coastal ecosystems such as mangroves, salt marshes, seagrass meadows, and more recently macroalgae.[3,4] These coastal blue carbon ecosystems (BCEs) have a disproportionally large C sequestration capacity. They are thought to contribute to over 60% of the total organic C buried in the global ocean sediments that, however, are only 0.5% of the seafloor area.[5] The annual C sequestration rate per area of BCEs is projected to reach 0.22 Gg C km^{-2}, which is tens to hundreds of times that of stable terrestrial forest ecosystems. The extremely high ability of BCEs to sequester large amounts of atmosphere CO_2 makes their conservation and restoration of BCEs as an important NbS for climate adaptation and mitigation.[6] Moreover, BCEs also provide many other ecosystem services, such as habitat provision, water purification, water conservation, flood and storm protection, shoreline stabilization, etc.

Coastal wetlands are typically distinguished as regions with slow organic matter (OM) decomposition followed by high rates of primary productivity.[5] As a result, scientific interest has taken off in past years because of the impact of these systems play on the global cycle as a large "Blue C" sinks.[1,6,7] Carbon sequestration in BCEs occurs primarily through the vertical sedimental C burial plus the net lateral C flux of dissolved inorganic C (DIC), dissolved organic C (DOC), and particulate organic C (POC) by tidal flow (Figure 1). Sediment C in BCEs is partially consumed by microbial processes. Part of the microbial waste products are flushed out of sediments via tidal movements, and part remains in the sediments. The long-term anaerobic environment slows down organic C decomposition, enabling vertical C accumulation.[8] Unlike the soils in terrestrial ecosystems, the sediments in BCEs do not become saturated with C as they accrete vertically under the impact of rising sea levels.[5] The rate of sediment C sequestration can be maintained as long as the soil accretion continues to keep pace with sea level rise.[9] The longevity of tidal C sinks thus has a more significant potential to mitigate climate change over longer periods. As a result, coastal wetlands are regarded as a significant player in the global C cycle, with large C reservoirs that continue to accumulate, ideal for climate change mitigation strategies.

While much research has focused on quantifying C stocks and vertical fluxes into sediments, the inorganic C input into the ocean driven by tidal flushing in

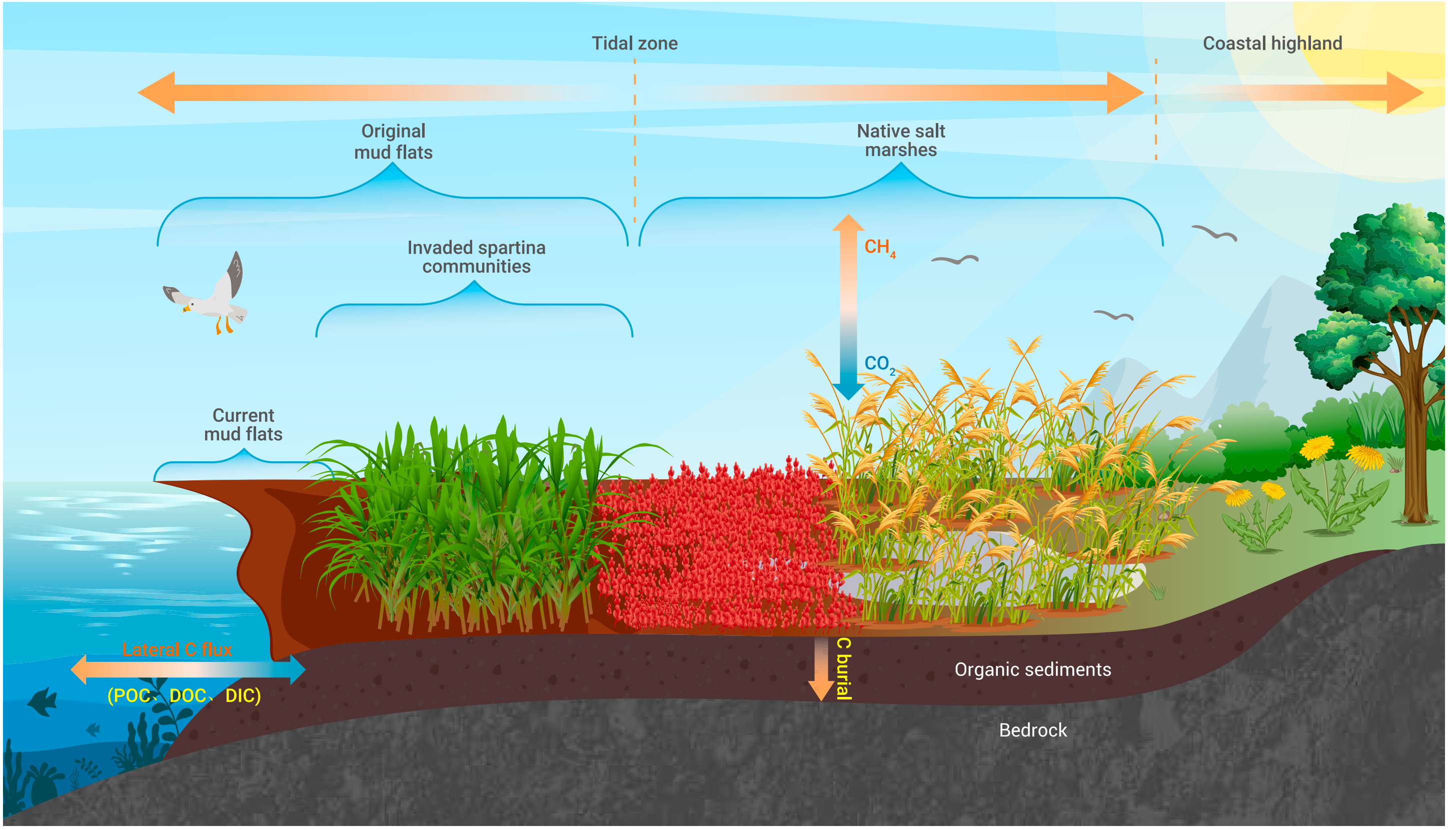


Figure 1. The vertical and lateral C flux of tidal wetlands in China modified from Wang et al.[66] MEHWS, mean extreme high spring water; MHW, mean high water; MLW, mean low water.

coastal wetlands (Figure 1) can exceed C burial.[10] Thus, the actual annual C sequestration capacity of coastal wetlands seems to exceed by far the rate estimated by the vertical C sequestration.[11] In addition, the presence of sulfate-rich seawater in tidally influenced BCEs partially suppresses the production of methane (CH_4),[12,13] a more powerful greenhouse gas (GHG) with a 100 year warming potential, 25 times that of CO_2. With rapid C sequestration and minimal CH_4 production, the BCEs are of high value for mitigating GHG emissions and are increasingly being considered as a nature-based climate solution for C removal.[5,6,14]

This paper reviews whether BCEs in China can contribute to climate mitigation. We first discuss the key mechanisms driving C sequestration in BCEs. Global- and China-scale estimates of blue C stocks, C burial, and lateral C flux are also described. Then, we discuss the climate change and human disturbance effects on BCEs. The potential practices and technologies that can be considered in China to improve BCE C sequestration, including their constraints and feasibility, are outlined, focusing on conservation and restoration opportunities in China. The role of BCEs in helping China achieve "C Neutral" by 2060 is examined, and future directions are suggested to improve blue carbon estimates on distribution extents, stocks, sequestration, and mitigation potentials.

THE CONTRIBUTION OF CHINA TO GLOBAL BCE C STORAGE

Mangroves

Carbon within BCEs is mainly stored in sediments rather than biomass.[15] Previous research has shown that most mangrove plant-fixed C is stored in the sediments, and only approximately 17% is stored in the biomass.[16] Carbon sequestration in mangroves is closely related to local biogeochemical and ecological processes.[17,18] Therefore, the C stored in mangrove sediments varies significantly among different geographic locations and forest types. Generally, the C stocks are higher in tropical areas than subtropical mangroves. For example, the estimated C stock was 662 Mg C ha^{-1} in Mexico,[19] 572 Mg C ha^{-1} in Indonesia, and 1,059 Mg C ha^{-1} in Malaysia,[17] while the C stock of mangroves in China is estimated to be 355.35 Mg C ha^{-1},[16] and only 119.3 Mg C ha^{-1} in Japan.[20] On a global scale, the average mangrove forest C stock range from 650 to 749 Mg C ha^{-1},[21,22] based on different estimation methods. Different latitudes and environmental conditions may create different soil physicochemical conditions in mangrove forests and thus affect the sediment C decomposition rates.[23] Globally, there are 13.7 million ha of mangrove forest area within tropical, subtropical, and warm temperate coastal zones, and the total C stocks in mangroves range between 3.7 to 11.7 Pg C.[6,24,25]

As the most dominant ecosystems of subtropical and tropical coasts, mangroves are mainly distributed in the southeastern region of China (Figure 2A), especially in Fujian, Guangdong, Guangxi, and Hainan provinces, accounting for more than 84.2% of the total mangrove area in China.[26] According to a remote sensing report,[27] China's mangroves covered a total area of 25,900 ha in 2015 (Table 1). The National Forestry and Grassland Administration's data indicated that Chinese mangroves had recovered rapidly over the past decade, reaching an area of 28,900 ha in 2020, with more than 7,000 ha that were newly built and recently restored. The extent of mangroves varies in different provinces. In Hainan, 55% of the local mangroves were converted to aquaculture ponds from 1996 to 2009, leaving the remaining mangroves fragmented throughout the region.[28] In Guangdong province, mangroves covered 9,305 ha in 1985, 9,556 ha in 1995, 6,793 ha in 2005, and 9,700 ha in 2015, respectively, with the changes attributed to natural and anthropogenic factors.[29] The mangrove area of Guangxi province increased significantly, and the area of mangroves exceeded the 1973 level by 2020.[26] China lost about 50% of its mangrove forests from 1950 to 2001 but, since 2001, the mangrove forest area has increased by nearly 2% $year^{-1}$ due to protection and restoration.[30]

China's mangroves have an average of 59.4 Mg C ha^{-1} stored in total biomass, 157.84 Mg C ha^{-1} stored in soils to a depth of 1 m, for a total C storage of 217.24 Mg C ha^{-1} and 72.65% of C stored in the mangrove sediment.[31] Multiplying the area, mangroves in China store approximately 6,918 Gg C (Table 1; Figure 2A). Alongi[32] estimated that global mangrove C storage was 933 Mg C ha^{-1}. Kauffman et al.[24] reported that Asian mangroves average 772.6 Mg C ha^{-1} with 294.8 Mg C ha^{-1} stored in the top 1 m of soil. Of the 38 mangrove species found in China (primarily *Avicennia marina*, *Kandelia candel*, and *Aegiceras corniculatum*), about 70% had tree heights not more than 3 m, and more than 90% had tree heights not more than 5 m.[33] These data indicate that China's mangroves have relatively lower biomass C storage compared with the global averages.

Table 1. The distribution, average C stock, C storage, and C burial of mangroves in China

Province	Area (ha)	Soil C stock (Mg C ha^{-1})	Soil C storage (Gg C)	Biomass C stock (Mg C ha^{-1})	Biomass C storage (Gg C)	Total C storage (Gg C)	C burial (Gg C a^{-1})
Zhejiang	106	103.86	11	76.11	8	19	0.21
Fujian	827	103.86	86	76.11	63	149	1.6
Guangdong	9,205	142.13	1,308	66.45	612	1,920	17.86
Guangxi	11,251	255.59	2,876	94.57	1,064	3,940	21.83
Hainan	3,630	159.10	578	42.86	156	733	7.04
Hong Kong	104	142.13	15	66.45	7	22	0.2
Macao	13	142.13	2	66.45	1	3	0
Taiwan	736	103.86	76	76.11	56	132	1.43
Total	25,872		4,951		1,966	6,918	50.17

The area data sourced from Mao et al.,[27] the biomass and soil C stock sourced from Fu et al.,[38] C burial data sourced from Wang et al.[65] 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

The reason may be the young age of recently restored mangroves, the higher latitude, and more frequent human disturbance.[16] Many studies have investigated the relationship between latitude and mangrove C stock,[34,35] and found that the lower latitude usually had higher mangrove C stocks,[35] the relatively high latitude thus should be the most important cause of the low biomass C stocks in Chinese mangroves.

Salt marshes

The global area of tidal marshes has been estimated to be in the range of 2.2–40 million ha.[6] The most complete and comprehensive assessment suggests that there are about 6.4 million ha tidal marshes at the global scale.[36] The data were accepted by the United Nations Environment Program World Conservation Monitoring Center. However, the mapping data in some underdeveloped countries are incomplete or even missing.[6] Salt marshes are among the most productive ecosystems, which store substantially more C in soils (317 ± 19 Mg C ha^{-1}) than in plant biomass (9 ± 4 Mg C ha^{-1}) with a total C stock of 334 ± 4 Mg C ha^{-1}.[18] Multiplying stocks by their global area, the global C stock in salt marshes is about 1.84 Pg C.[36]

Salt marshes are the largest type of coastal BCEs in China. However, the total distribution of salt marshes in China is actually of great discrepancy.[33] The recent national scale remote sensing of wetlands found that the salt marsh area was 297,936 ha.[27] The Chinese government investigation between 2009 and 2013 through the Second National Wetland Resources Report in 2014 suggested that there were 343,000 ha salt marshes in China, while Hu et al.[37] based on Sentinel-1 time-series data reported the total area of Chinese salt marshes as 127,477 ha (Table 2). In conclusion, the area of salt marshes in China ranged from 127,477 to 343,000 ha. In view of salt marsh C storage, Fu et al.[38] calculated that the soil C storage and the C stock (±95% CI) of salt marshes in China were 7.5 ± 0.6 Tg C and 81.1 ± 9.1 Mg C ha^{-1}, respectively. Combined with the total area reported by Mao et al.,[27] we calculate the total C stocks of Chinese salt marshes of 24.86 Tg C (Table 2).

Seagrass

Estimates of global seagrass spatial distribution ranged from 17.7 to 60 million ha in previous literature,[39,40] but recent modeling suggests that the mapped area of seagrass ranged from 16.04 to 26.66 million ha with moderate to low or high confidence, respectively.[41] Based on different mapping techniques, the potential area for seagrass occupation globally ranges from 16.47 to 43.20 million ha.[42] There have been several studies conducted to estimate the global seagrass C storage. Fourqurean et al.[43] estimated that seagrass ecosystems could conserve up to 19.9 Pg C, but 4.2–8.4 Pg C utilizing a more conservative method. A recent review estimated that the C storage in seagrass ecosystems ranged from 1.73 to 21 Pg C.[6] The estimated C storage of seagrass ecosystems ranged in an order of magnitude due to (1) the great uncertainties in the seagrass distribution mapping, (2) the changing status of seagrass coverage over time and the difficulties in accurately identifying seagrass in some certain water depth and clarity,[44] and (3) the imbalanced investigation region,[45] showing that most data were collected from North America, Western Europe, and Australia, with limited data in South America, Africa, and tropical Indo-Pacific.[43]

Nowadays, the spatial distribution of seagrass ecosystems in China contains a large discrepancy because seagrass meadows are submerged by seawater most of the time, and optical instruments cannot distinguish the difference between seagrass and the surrounding seawater. According to Zheng et al.,[46] the total area of seagrass ecosystems in China is around 9,000 ha. Recently, however, some new regions of seagrass beds have been discovered in Sansha City, Hainan Province, Bohai Sea, and Liaodong Penisula.[47–50] Based on the distribution characteristics, seagrass ecosystems in China can be divided into South China Sea, the Yellow Sea, and Bohai Sea.[51] The total area of seagrass ecosystems in China's coastal zone varies greatly due to the different mapping methodologies, and here we sum up these data and report a total area of 14,169 ha (Table 3). However, a unpublished national survey reveals that the total area of seagrass beds in China is 26,495 ha. At present, a unified remote-sensing geographic information system database is of particular importance for C sink estimation and long-term monitoring of seagrass ecosystems.

Previous studies found that the C storage of seagrass ecosystems in China is around 0.95 Tg C,[52] while recent studies found that the seagrass ecosystems contain around 1.6 Tg C and the C burial amount was around 6 Gg C year^{-1}.[38] However, the C storage and burial rates vary significantly among study sites. Previous research has found that the primary productivity of the seagrass beds distributed in Sanggou Bay was 543 g C m^{-2} year^{-1}.[53] Another study has found that the C accumulation rate in the lagoon and northern coast of Dongsha Island are approximately 34.5 and 31.2 g C m^{-2} year^{-1}, respectively.[54] Recent studies have discovered that the total organic C storage seagrass beds in Guangxi Province were 26.72 Gg C[55] Moreover, the total organic C stock in newly discovered seagrass beds in Hainan Island was around 1.31 Gg C, and the total organic C stock across Hainan Island was 40.9 Gg C[56] Here, we estimate a total seagrass meadow C storage of 1,396 Gg based on the updated seagrass meadow area (Figure S1; Table 3).

Tidal flats

Tidal flats are also an important type of coastal ecosystem, which mainly includes mudflat, sandy beach, and bedrock coast, as these areas have a strong C burial capacity. The global area of tidal flats reaches up to 12.8 million ha.[57] The average C accumulation rate of tidal flats is 129.8 g C m^{-2} year^{-1}, with the top meter sediments containing on average 86.3 Mg C ha^{-1}.[58] Globally, tidal flats can bury 6.8 Tg C per year and can store 0.9 Pg C in the top meter sediment.[58]

China's tidal flats occupy a large area, even exceeding the total area of salt marshes and mangroves. Indeed, the tidal flats in China are dominated by mudflats, characterized by a high burial rate of sediment and strong C sequestration potential. The total area of tidal flats in China ranged from 0.24 to 1.1 million ha based on different mapping sources.[27,57] The unvegetated tidal flats occupied 87% of the entire tidal area in China, and stored 78.07 Tg C (top 1 m soils), accounting for nearly 80% of the C deposited in the entire coastal tidal area.[59] By

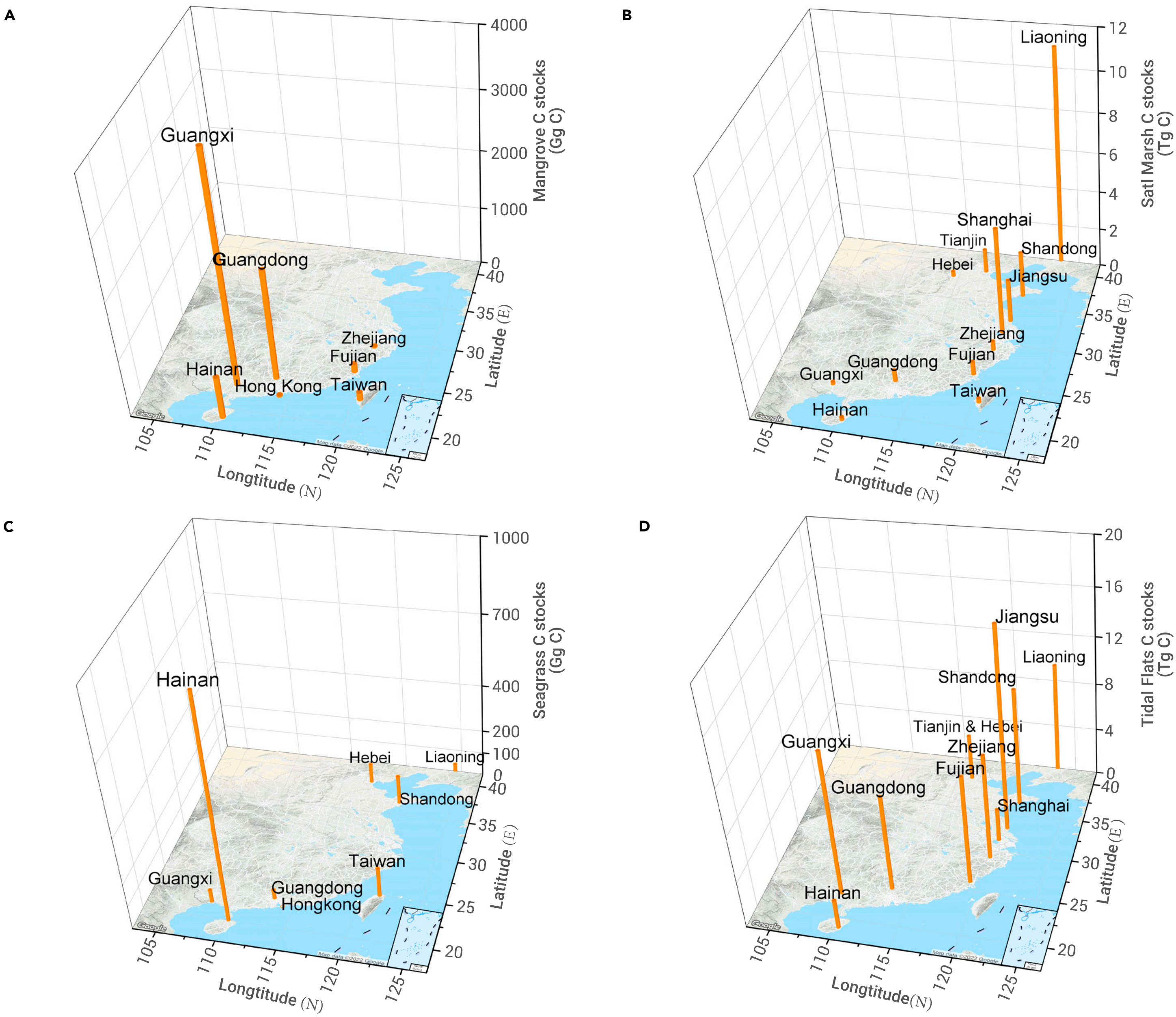


Figure 2. The provincial total C stocks of mangroves, salt marshes, seagrass, and tidal flats in China The provincial total C stocks of mangroves (A), salt marshes (B), seagrass (C), and tidal flats (D) in China. The data source refers to Tables 1, 2, 3, and 4. 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

combining the reported area and average C stock data, we estimate that the total C storage of tidal flats in China ranges from 27 to 85 Tg C (Table 3; Figure 2D).

THE CONTRIBUTION OF CHINA TO GLOBAL BCE C BURIAL

Mangrove

The C sequestration of BCEs is mainly through vertical sedimental C burial (Figure 1). Many studies have estimated the global organic C burial rates using different approaches for mangrove forests. The first C burial rates in mangroves were compiled by Twilley et al.,[60] Alongi,[32] and Breithaupt et al.[61] estimated the global mangrove C accumulation to be 29 and 2.6 Tg C year^{-1}, respectively. Rosentreter et al.[62] estimated the mangrove C burial rates of 31.3 Tg C year^{-1}. Recent studies suggested the range of mangrove C accumulation rates was 18.4–34.4 Tg C year^{-1} [5,32,61,63] based on the total mangrove area of 13.7 million ha.[64] However, most of the above-reported values were based on the simple multiplication of the C burial rate with global or local mangrove area, creating biases against undersampled tropical mangroves. A spatially explicit database of C burial rates is needed for Tier 2 estimation of global mangroves. To fill this gap, Wang et al.[65] established a comprehensive global tidal wetland C burial database, which includes some regions that have never been reported before, such as Africa, and found that the C burial by global mangroves reached up to 41 Tg C year^{-1} (Figure S1A).

Although the C storage of mangroves in China is lower than that in other countries, the C burial rate is relatively high when compared with other regions as a result of the high sedimentation accretion rates.[66] Ren et al.[67] calculated the average C burial rate for young *S. apetala* plantation, which was 750 g C m^{-2} year^{-1}. Meng et al.[33] estimated the mangrove C burial rate in China was 226 g C m^{-2} year^{-1}. These values are high compared with the global average of 170–190 g C m^{-2} year^{-1}.[65,68] We estimated that the total C burial amount of mangroves in China (Figure S1) is about 0.05 Tg C year^{-1},[65] which is close to other studies,[38] but far less than that of salt marshes in China, mainly due to the currently small areas of mangroves.

Salt marsh

The capacity of long-term C burial by salt marshes primarily depends on the balance between inputs (OM produced *in situ* and *ex situ*) and outputs.[69] The high C burial rates of salt marshes (164–218 g C m^{-2} year^{-1}) are about 40 times that of terrestrial ecosystems, acting as important C sinks and mitigating the impacts of sea level rise on the coastline.[5,65,70] With the spatially explicit database of

Table 2. The distribution, average C stock, C storage, and C burial of salt marshes in China

Province	Area (ha)		Soil C stock ($Mg\ C\ ha^{-1}$)	Soil carbon storage (Tg C)		Soil burial ($Gg\ C\ a^{-1}$)	
	Low	High		Low	High	Low	High
Liaoning	4,481	97,473	112.6	0.50	10.98	7.48	162.78
Hebei	152	10,347	25.1	0.00	0.26	0.25	17.28
Tianjin	204	18,969	65.2	0.01	1.24	0.34	31.68
Shandong	20,206	42,134	56.6	1.14	2.38	33.74	70.36
Jiansu	30,818	46,598	48	1.48	2.24	51.47	77.82
Shanghai	32,956	60,266	91.7	3.02	5.53	55.04	100.64
Zhejiang	25,016	7,660	60.7	1.52	0.46	41.78	12.79
Fujian	11,025	5,121	137.3	1.51	0.70	18.41	8.55
Guangdong	372	5,361	110.9	0.04	0.59	0.62	8.95
Guangxi	2,249	898	138.5	0.31	0.12	3.76	1.50
Hainan	–	1,567	92.6	–	0.15	–	2.62
Taiwan	–	1,541	135.8	–	0.21	–	2.57
Hong Kong	–	2	110.9	–	0.00	–	–
Macao	–	<0.1	110.9	–	0.00	–	–
Total	127,477	297,936	54.10	9.55	24.86	213	498

The area data sourced from Mao et al.[27] and Hu et al.,[37] the soil C stock sourced from Fu et al.,[38] C burial data sourced from Wang et al.[65] 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

C burial rates in tidal wetlands, Wang et al.[65] reported that the C burial by global tidal marshes was 12.6 Tg C $year^{-1}$ (Figure S1B).

The C burial rates of salt marshes in China were estimated at 7–955 g C m^{-2} $year^{-1}$ with mean and median values of 201 and 154 g C m^{-2} $year^{-1}$, respectively.[38] Fu et al.[38] estimated that the C burial rate of salt marshes in China was about 0.16 Tg C $year^{-1}$. Our previous studies estimated that the C burial of salt marshes in China was 1.19 Tg C $year^{-1}$.[65,66] These data were greater than the previous estimate of 0.16–0.75 Tg C $year^{-1}$, mainly because the salt marsh area (548,000 ha) used was larger than other data sources.[65] If using a relatively conservative wetland area (297,900 ha) to calculate these values (Table 2), we estimated the annual C burial capacity of salt marshes in China is about 0.50 Tg C a^{-1}.[66]

Seagrass

Previous studies showed that the rate of C burial in seagrass meadows was approximately 83 g C m^{-2} $year^{-1}$,[71] and the global C burial rate of seagrass ecosystems varied from 27 to 44 Tg C $year^{-1}$, which was an important component of total C burial in the ocean.[72] Kennedy et al.[73] using paired C isotopic composition data, estimated that the C burial rate was 48–112 Tg $year^{-1}$. However, the organic C only stands for a small part (2%–3%) of the buried material within seagrass sediments,[43] with the remaining being the inorganic C.[74] The inorganic C burial in seagrass ecosystems is estimated to be 15–62 Tg C $year^{-1}$.[75] The high C burial rate in seagrass ecosystems contributes to seabed elevation and therefore acts as a buffer against sea level rise. The net effect of organic versus inorganic C burial in seagrass beds is difficult to assess without deep biogeochemical insight into the processes driving the accumulation and cycling of sediment carbon. In China, the seagrass meadow C burial rate ranged from 5 to 379 g C m^{-2} $year^{-1}$, with the total burial C up to 10.47 Gg C $year^{-1}$ (Table 3).

Tidal flats

Low C densities (<0.01 g cm^{-3}) were found in China's tidal flat sediments,[59] but the C sequestration rates ranged from 35 to 361 g C m^{-2} $year^{-1}$, which are comparable with the nearby salt marshes and mangroves rates.[59,66] Based on the most conservative coastal mudflats distribution data and C burial rate of nearby salt marshes and mangroves, the lower limit of C burial rate of coastal tidal flats in China is estimated to be 0.42 Tg C $year^{-1}$, which is much higher than that of mangroves and second only to salt marshes in China.[66] Here, we combined the tidal flats C burial rate[59] and its different area sources,[27,57,59] and calculated the total C burial in China tidal flats ranging between 0.28 and 1.5 Tg C $year^{-1}$ (Table 4). However, the future C sequestration rates of these tidal flats are facing uncertainties under the pressures of reduced fluvial sediment loads from major rivers.

VERTICAL GHG EMISSIONS AND LATERAL C FLUX OF BCEs

Methane emissions

BCEs are greatly divergent in the GHGs emissions, owing mostly to their locations spanning from temperate to tropical climates and regulated by irregular tides. Methane (CH_4) emission in BCEs was regarded to be considerably low by the traditional concept, primarily because of the presence of sulfate.[76] However, previous literature data indicated that CH_4 emission from sediments varies greatly due to the large differences in soil organic matter (SOM) among sites, vegetation types, salinity gradients, temperature, and the availability of terminal electron acceptors.[77–79] Some studies indicated that the anthropogenic nutrient loading (total organic C, N, and P) from the drainage triggers higher CH_4 emission rates.[80,81] Ma et al.[80] reported that heavy metal pollution significantly increased the emission of CH_4 up to 180 μg g^{-1} day^{-1}. The CH_4 emissions can also be emitted through the diffusion of water from creeks, but that is generally smaller than emissions from the sediments, because the CH_4 can be oxidized by methane-oxidizing bacteria in the surface water before it reaches the atmosphere. In addition, recent studies reported that CH_4 could be emitted through the stems and leaves in the mangrove due to the aerobic conditions.[82] Rosentreter et al.[62] showed that high CH_4 evasion rates have the potential to partially offset blue carbon burial rates in mangrove sediments on average by 20%. Thus, the CH_4 fluxes in mangrove ecosystems cannot be neglected when constructing the C budgets and estimating the C sequestration. GHG emissions of salt marshes in China were estimated at rates of 23.6–986 μg CH_4 m^{-2} h^{-1} and 1.58–110 μg N_2O m^{-2} h^{-1}, respectively.[83–85] At the same time, GHG emissions of salt marshes in China increased from high to low latitudes, indicating the high dependence on climatic temperature.[85] Based on the reported data,[62] we estimated the CH_4 emission of mangroves in China would be 10 Gg C $year^{-1}$ (Figure 3), while further work is needed to calculate the methane emission in other BCEs (especially salt marshes and tidal flats) in China.

Table 3. The distribution, average C stock, C storage, and C burial of seagrass meadows in China

Province	Area (ha)	C stock (Mg C ha^{-1})	Total C stock (Gg C)	C burial rate (g C m^{-2} year^{-1})	Total C burial (Gg C year^{-1})
Liaoning	892	33.6	30.0	202	1.8
Hebei	2,917	27.6	80.5	24	0.7
Shandong	1,352	90.9	122.9	379	5.12
Guangdong	1,039	37	38.4	45	0.47
Guangxi	942	56.4	53.1	202	1.9
Hainan	5,838	162.5	948.7	7	0.41
Taiwan	1,186	103.4	122.6	5.05	0.06
Hongkong	4	36.2	0.1	202	0.01
Total	14,169	90.75	1,396.4	–	10.47

The area data sourced from previous reports,[46,47,49,50,55] the C stock sourced from Fu et al.,[38] C burial data sourced from Fu et al.[38] 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

Lateral C flux

Besides the better-understood vertical C burial and GHG emission, BCEs also furnish C and nitrogen to adjacent coastal waters through lateral tidal flow.[11] The inorganic C input into the ocean by lateral tidal flow from coastal wetlands (also referred as outwelling) can exceed the sedimentation of organic C.[10,86] For instance, mangrove outwelling contributes ~10% of the terrestrially derived DOC to the oceans.[87] The export of DIC and alkalinity was approximately 1.7 times higher than burial as a long-term C sink in a subtropical mangrove system.[86] By using the global organic C burial rate of 41 Tg C year^{-1},[65] then the mangroves C sink associated with DIC and alkalinity export may reach 69.7 Tg C year^{-1}. Besides DIC, POC and DOC exports to the coastal ocean are also significant in terms of the C budget (Table 5), and combined this may account for an atmospheric C sink similar to the burial.[86] The rates of POC export to adjacent tidal waters are three times greater in mangroves than in salt marshes, DOC export is more than twice the rate in mangroves than in salt marshes, and DIC export is nearly three times greater in mangroves than in salt marshes.[18] Global upscaling exercises based on short datasets on a few sites imply that the mangrove DIC, DOC, and POC outwelling exports approach 190 Tg C year^{-1}.[11] Here, we synthesized the reported lateral flux from previous studies (Table 5): mangroves on average exported 588, 146, and 98 g C m^{-2} year^{-1} and salt marshes exported 213, 97, and 9 g C m^{-2} year^{-1}as DIC, DOC, and POC, respectively. Based on these global average values, the lateral flux of mangroves and salt marshes in China are 215 and 951 Gg C year^{-1}, respectively (Figure 3).

These large DIC, POC, and DOC exports to the coastal ocean are significant from a C budget perspective.[86] However, it remains unclear whether the exported C is stored in the ocean over long time scales, or returns to the atmosphere. Part of the POC and DOC will be oxidized back to DIC in coastal waters, and some parts can be transformed into refractory DOC (RDOC) by the microbial C bump.[88] The DIC is made up of both carbonate alkalinity (HCO_3^- and CO_3^{2-}), which can be stored in the ocean over thousands of years, and CO_2, which returns to the atmosphere.[11] Resolving the relative contribution of alkalinity and CO_2 to DIC is essential to understanding the contribution of outwelling to blue carbon sequestration. The ratio of alkalinity to DIC is driven by complex biogeochemical processes that are difficult to quantify. Alkalinity seems to contribute to most of the DIC exports in mangroves and saltmarshes in the few cases when alkalinity was quantified.[86,89] If this proves correct as more observations are performed, the export of DIC and alkalinity results in a long-term atmospheric C sink and should be incorporated into the blue C paradigm to assess its role in sequestering C and mitigating climate change.[86] RDOC is a huge and stable carbon reservoir in the ocean equivalent to total amount of CO_2 in the atmosphere.[90] The contribution of RDOC to the lateral C flux in BCEs is influenced by environmental conditions and thus allows manipulations to pursue maximum outputs of C sequestration.[3]

CLIMATE CHANGE AND HUMAN DISTURBANCES ON BCEs

Climate change

Climate change is exerting a great and continuous influence on BCEs. Coastal wetlands face numerous challenges due to elevated atmospheric CO_2 concentration, sea level rise, air and water temperature rise, and changes in precipitation patterns and storm events.[91] Moreover, climate change influences coastal wetland distribution and function, and affects C cycling and GHGs emissions from those ecosystems.[92,93] The following section focuses on the influence of climate change on productivity, soil C burial rate, and GHG production in BCEs.

Ecosystem productivities play an important role in SOM accumulation in coastal wetlands, and canopy respiration is the largest component of CO_2 exchange between the ecosystem and atmosphere.[94] Increased temperature will enhance plant productivity in coastal wetlands until a threshold is reached.[94] Generally, in tropical forests with abundant water supply, the temperature may be more important than parameters related to moisture in moderating coastal wetlands productivity, while availability from tides and rainfall seem to play a more critical role in semi-arid climates.[95,96] Annual litter fall has typically been used to assess the mangrove productivity, showing latitudinal gradients, with greater litterfall at latitudes lower than 10° and lower litterfall in latitudes higher than 30°.[60,63] For instance, the pattern of litterfall along latitudes was in line with the changes in sediment organic C stock in global mangroves.[25] Salt marsh productivity also possessed a north-south gradient that follows the solar energy inputs at 0.2%–0.35% net conversion efficiency.[97] Enhanced precipitation can increase nutrient inputs to estuarine systems, flood length, and reduce pore water salinity in coastal wetland soils, which may lead to tree growth.[35] Conversely, coastal wetlands in arid or semi-arid climates are likely affected by decreased rainfall and may have lower productivity, particularly due to higher salt stress.[98] Consequently, precipitation patterns driven by climate change will exert an influence on both ecosystem gross productivity and respiration, altering GHG emissions as well as the blue C budget. Sea-level rise will also likely affect photosynthesis and therefore productivity in coastal wetlands, particularly by moderating nutrient availability and inundation frequency to partially control physico-chemical properties such as pore water salinity and soil redox conditions.[99] Ouyang et al.[100] recently reported that litter carbon sinks in BCEs will increase due to higher increase in litter C production than in decomposition by 2100 compared with 2020 due to climate change, which highlights that BCEs will play an increasingly important role in future climate change mitigation.

Carbon burial can be highly variable, significantly influenced by the soil accretion rate and therefore the tidal regime and surface elevation. Carbon burial rates in coastal wetlands have therefore been greatly influenced by the changing sea level.[9] Although, in some places, mangroves keep pace with sea level rise through vertical surface elevation change,[101] some mangroves cannot and therefore habitats may be lost. Over relatively long periods of time, coastal wetlands have to migrate with the intertidal zone and move to new areas to maintain their physiological needs,[102] which influences C burial rates. Current sea level rise is inducing a landward migration of coastal wetlands that will influence C burial in their soils, modifying C stocks and thus GHGs production and emission. Coastal wetlands may directly influence soil accretion processes through root system development, which can also be altered under climate change.[103] Coupled with sea level rise, climate change factors, such as increases in rainfall and temperature, greatly influence subsidence, root development, and sediment inputs from watersheds, all of which impact burial rates in coastal ecosystems.[104] Moreover, a recent study reported that plant trait evolution and/or plant adaption can alter coastal wetland resilience to sea level rise,[105] suggesting that plants can evolve at a pace that feeds back on ecosystem processes.

Besides organic-rich soils, coastal wetlands are usually characterized by low production rates of GHGs, CO_2, or CH_4.[106] In the soil, GHG production through microbial processes depends on many parameters, including oxygen availability, soil and water temperatures, soil water content (SWC), grain size, porosity, redox potential (Eh), salinity, pH, and OM quality.[106,107] Enhanced temperature will stimulate OM decomposition, and therefore GHG production. Increased temperature is positively correlated with C mineralization in mangrove soils.[108] For example, CH_4 emissions from salt marsh soils were found to correlate with both air and soil temperatures, indicating that methanogenic bacteria are better adapted to warm soil conditions than methanotrophic bacteria.[109] At the water-air interface, CO_2 production in mangroves was also found to increase significantly following

Table 4. The distribution, average C stock, C storage, and C burial of tidal flats in China

Provinces	Area (ha)		C stock ($Mg\ C\ ha^{-1}$)	Total SOC storage (Tg C)		C burial rate ($g\ C\ m^{-2}\ year^{-1}$)	Soil C burial amount ($Gg\ C\ year^{-1}$)	
	Low	High		Low	High		Low	High
Liaoning	0.54	133,100	69.9	0.00	9.30	107.45	0.00	143
Tianjin and Hebei	8,305	73,200	53.6	0.45	3.92	146.02	12	107
Shandong	34,208	123,800	82	2.81	10.15	192.95	66	239
Jiangsu	6,277	291,400	59.4	0.37	17.31	153.84	10	448
Shanghai	10,981	37,800	73.7	0.81	2.79	91.51	10	35
Zhejiang	21,740	132,800	67.9	1.48	9.02	153.84	33	204
Fujian	28,285	122,300	75.74	2.14	9.26	139.38	39	170
Guangdong	34,807	108,500	75.1	2.61	8.15	74.55	26	81
Guangxi	69,732	59,800	207	14.43	12.38	74.55	52	45
Hainan	5,031	19,700	125.7	0.63	2.48	136.63	7	27
Taiwan	18,075	–	75.7	1.37	–	139.38	25	–
Hongkong	2	–	75.1	0.00	–	74.55	0	–
Macao	7	–	75.1	0.00	–	74.55	0	–
Total	237,450	1,102,400		27.1	84.8		280	1,499

The area data sourced from Mao et al.[27] and the global tidal flat map,[57,59] the soil C stock sourced from Chen et al.,[59] C burial data sourced from Chen et al.[59] 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

the increasing water temperature.[110] Changing precipitation patterns and sea level rise may influence litter and OM decomposition along with GHG production within coastal wetland soils, particularly through altering the SWC, the oxygen supply, and soil electron acceptor renewal.[35,96,107,111] However, microbial metabolisms can be stimulated under suitable moisture: SWC less than 10% constrains the normal metabolic activity in the soil, and droughts are likely to limit OM mineralization and hence GHG production.[112] Elevated atmospheric CO_2 concentrations can enhance C:N ratios of coastal wetland plant tissues, such as mangrove seedlings.[113] Generally, climate change may result in a more recalcitrant OM, inhibiting the decomposition process and GHG production. In addition, elevated CO_2 concentrations are causing ocean acidification, which may also influence microbial activity in coastal wetland soils.

Human disturbance

By far, the major cause of historical and contemporary coastal wetland loss is direct human modification.[114] During the 20th century, the conversion of wetlands into other land use reached about 25%–50% of the world's coastal wetlands.[5,115] Tidal marshes were among the earliest coastal wetlands to be substantially modified[116] because they dominated the temperate zone where industrialization took place. Global losses of tidal marshes and seagrasses have been estimated at around 1%–2%[117] and 1.5%[118] per year, although some conservation actions have led to a reversal of declining trends in seagrasses in certain regions.[119] Globally, the mangrove loss rate had declined to an average of 0.16%–0.3% per year between 2000 and 2012 from 0.99% in the 1980s, which shows the efforts related to mangrove conservation.[120] Even though mangrove soils alone have lost an estimated 30.4–122 Tg C due to land-use changes that occurred between 2000 and 2015, with Indonesia, Malaysia, and Myanmar contributing to more than 75% of these losses.[121] The main drivers of BCE loss vary across ecosystems and geographical regions, but usually include physical change (including ecosystem change and drainage), pollution, non-native species, and climate change.[115] For instance, developing countries are now converting coastal wetlands into other land uses at breakneck speed, replacing agriculture, aquaculture, and tourism and providing natural capital and ecosystem services to these systems.[117] In summary, BCEs not only store C but also provide benefits to both human beings and ecosystems, such as coastal protection and fisheries enhancement;[6] however, more work is needed on BCE conservation to maintain these benefits and improve sustainable development.

ECOSYSTEM SERVICES OF BCEs BEYOND C SEQUESTRATION

A broad definition of ecosystem services was provided by the Millennium Ecosystem Assessment that "Ecosystem services are the benefits people obtain from ecosystems."[122] With essential ecological functions, coastal BCEs are very productive and provide many benefits to society, such as habitat provision, climate regulation and stabilization, water purification, water conservation, flood protection, shoreline stabilization, source of substantial biodiversity, and atmospheric maintenance.[101,123,124] Besides, coastal wetlands provide goods, such as food, water, raw materials, and other products collected by local communities. Coastal wetland ecosystem value ranged from US$300 to $887,828 ha^{-1} $year^{-1}$, and US$193,845 ha^{-1} $year^{-1}$ on average.[125,126] It was also reported that a 32,346 ha reduction in coastal wetlands resulted in US$806 million $year^{-1}$ loss in ecosystem service.[127] Here, we summarize the primary ecosystem functions of BCEs, and listed their values (Table 6). In general, these ecosystem services are worth around $32,000 ha^{-1} $year^{-1}$.

Water conservation

Coastal wetlands play an important role in regulating water flow and reducing the impact of floods and storm surges. They act as natural sponges by absorbing and storing large amounts of water during periods of high flow, and then releasing it slowly. This helps to regulate the water cycle and reduce the risk of flooding in nearby coastal communities. Coastal wetlands also help to reduce the impact of storm surges by acting as a buffer between the land and the sea. In addition, coastal wetlands can also help to recharge ground water and maintain water quality in nearby rivers and estuaries. Coastal wetlands also help to improve water quality by filtering pollutants and excess nutrients before they reach the ocean.

Storm protection

As natural buffers, coastal wetlands are effective against storms, preventing damages from flooding from storm-tidal surges, rainfall, and high winds. Between upland terrain and the storm surge, a coastal wetland can diminish storm intensity.[128,129] A gain of 1 km^2 of coastal wetlands decreases a loss of an average US$12.00 million and a median US$1.70 million, suffering from storm damage related to specific events.[130]

Water purification

Coastal wetlands are water quality control systems, filtering chemicals and sediment out of the water before it is discharged into the ocean. Coastal wetland

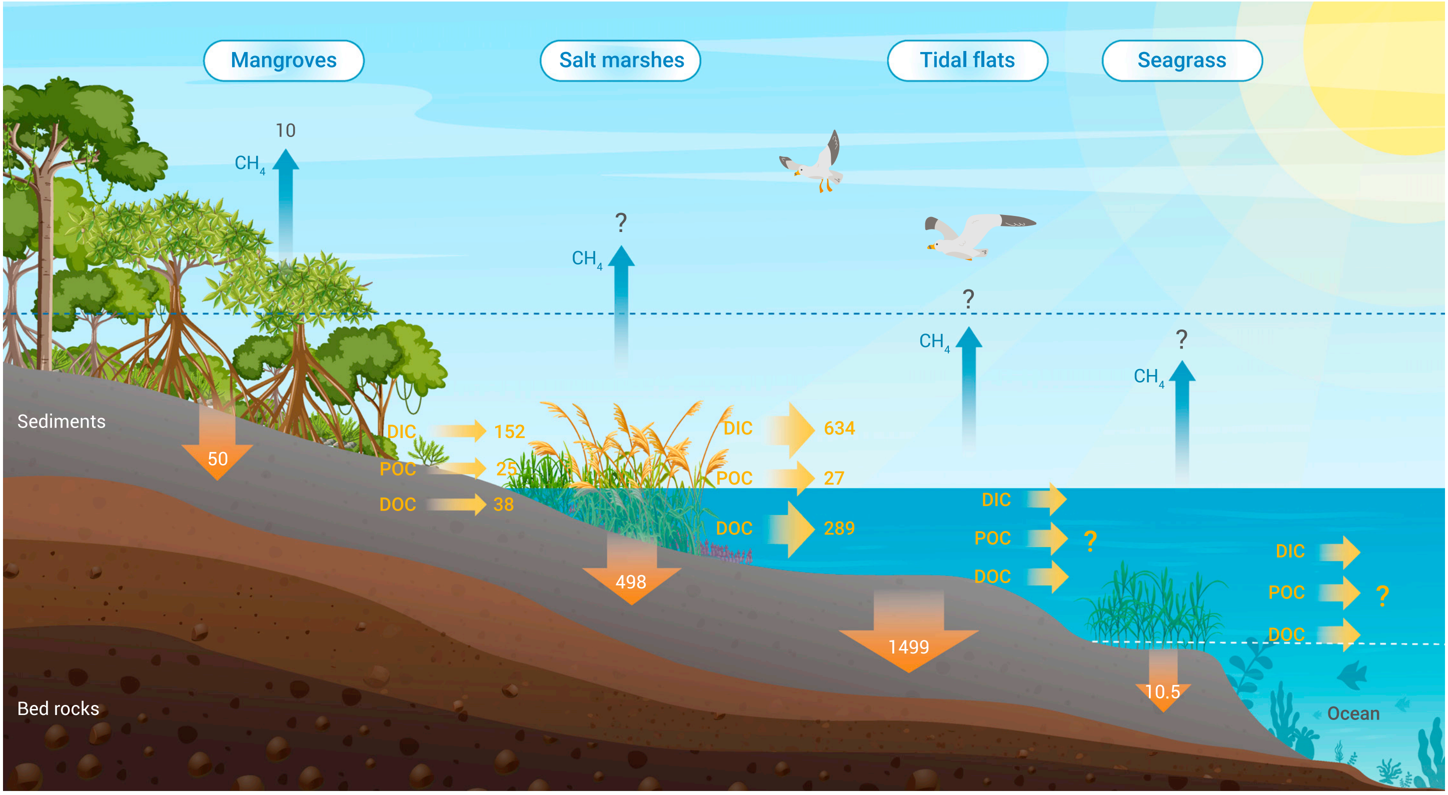


Figure 3. The vertical and lateral C flux of BCEs in China in units of Gg C year^{-1} 1 Tg = 10^3 Gg = 10^6 Mg = 10^9 kg = 10^{12} g.

plants and phytoplankton can fix nitrogen and phosphorus through photosynthetic processes. The water quality enhancement provided by coastal wetlands in Louisiana alone was estimated to cost US$99 to US$5,551 ha^{-1} on the basis of the type of treatment.[131]

Shoreline stabilization

With well-developed root systems, mangroves, miscanthus, and other wetland plants can hold the soil in place with their roots, absorb the energy created by ocean currents, and break up the flow of waves, currents, and runoff, which would otherwise degrade a shoreline and associated development. The value of seashore protection ranged from US$9,143 to US$30,757 ha^{-1} year^{-1}, which was mainly due to the affects related to coastal mudflats.[132]

Habitat provision/biodiversity

Providing habitat to thousands of flora and fauna, coastal wetlands are diverse and dynamic ecosystems with biodiversity that includes birds, fish, mammals, reptiles, and amphibians, maintaining from genetic diversity to species diversity to ecosystem diversity. Xie et al.[133] reported a factor of 2.5 for the contribution of biodiversity maintaining service in coastal wetlands compared with a factor of 1 for farmland ecosystems, resulting in a value of US$316 ha^{-1} year^{-1}.[134]

High productivity

With high biodiversity and high biomass, coastal wetlands provide a variety of socio-economic benefits (e.g., valued products) to people for human consumption, including fruits, fish, shellfish, meats, resins, timber for building, firewood, and so on.

BCEs AS A NbS FOR C NEUTRALITY IN CHINA

Conservation and restoration

Although the ecological and socioeconomic importance of BCEs has been greatly emphasized in the past, these systems are still rapidly being lost globally due to human disturbance as mentioned above. Globally, BCEs are expected to decline further due to various interconnected drivers, including deforestation, pollution, and climate change.[135] The rapid loss, fragmentation, and degradation of BCEs create strong incentives for their restoration across their natural range to replace lost habitat and ecosystem services.[13,14,136,137] Generally, the conservation of remaining BCEs is much more cost-effective than restoration as this requires substantial investment. However, restoration can earn more anthropogenic C credits, which offer opportunities to develop market-based mechanisms that take advantage of existing C offsets frameworks.[6] There have been numerous efforts globally to conserve and restore BCEs. Almost 2,000 km^2 of mangroves have been planted over the past 40 years.[135] A recent meta-analysis concluded that mangrove restoration offers positive benefit-cost ratios ranging from 10.50 to 6.83,[136] suggesting that restoration is a cost-effective form of ecosystem management. The restoration of BCEs globally was estimated to mitigate up to 290 Tg C year^{-1} emissions, with 2.18 Mg C ha^{-1} year^{-1}.[138] In comparison with restoration, conservation current BCEs will be a much more cost-effective pathway to C neutral.[138] The avoided C emissions by protecting threatened BCEs may reach up to 89 Tg C year^{-1}.[138] By using more recent lower estimates of ongoing BCE loss rates, Bossio et al.[139] estimated the conservation and protection of BCEs would avoid the soil C emission of 197.47, 133.78, and 77.43 Mg C ha^{-1} over the expected loss of 0.05 Mha year^{-1} mangroves, 0.08 Mha year^{-1} salt marshes, and 0.45 Mha year^{-1}seagrass, respectively. Overall, the conservation and restoration of BCEs have substantial potential to increase the C sequestration and contribute to an NbS for climate change.[1,6,66,138,140]

In China, coastal wetlands have experienced a sharp decrease in the area since the 1950s. An et al.[141] reported that the cumulative loss of the coastal wetland area is around 22,000 km^2 since 1949, about 51.2% of the total area of coastal wetlands in China. The total mangrove area has lost from 48,650 ha in the 1950s to 25,900 ha in 2015. China aims to restore the 48,650 ha of mangrove forests in the next 10 years.[38] If this is well conducted, there will be an additional 106 Gg C year^{-1} C sequestration in these mangroves. Besides mangroves, China has a large area of salt marshes and unvegetated tidal flats.[66] Unlike mangroves, the restoration of salt marshes has not been well planned. In fact, salt marshes in China have suffered great losses since the 1950s without experiencing any appreciable recovery. The area loss rate is estimated to have been relatively stable during the 1950s–1980s (28,893 ha year^{-1}) and 1980s–1990s (22,279 ha year^{-1}) then to have declined to 8,928 ha year^{-1}in the 1990s–2000s, with a total of 22.5–95.7 Tg C emitted throughout the 70 years,[38] equivalent to 321–1,367 Gg C year^{-1}. The rapid coastal development in China during the past decades also resulted in a loss of over 23,000 ha year^{-1}of tidal flats reclaimed for aquaculture, agriculture, salt pans, and urban expansion from the 1950s,[59,142] leading

Table 5. The reported coastal wetland lateral C flux worldwide

S.N	Site	Vegetation	DOC (g C m^{-2} $year^{-1}$)	DIC (g C m^{-2} $year^{-1}$)	POC ;(g C m^{-2} $year^{-1}$)	References
1	East and west coast marshes, USA	salt marsh			9.1	Ganju et al.[182]
2	US pacific coast	salt marsh	9.7	96		Bogard et al.[183]
3	North Inlet, South Carolina, USA	salt marsh	73 ± 73	2,628 ± 2,000		Correa et al.[184]
4	North Carolina, USA	salt marsh	178	227		Czapla et al.[185]
5	Delaware, USA	salt marsh	40	168		Guimond and Tamborski[186]
6	Sage Lot Pond, USA	salt marsh		414		Wang et al.[10]
7	York River, VA Estuary	salt marsh		193		Neubauer and Andersen[187]
8	Duplin River, Sapelo Island	salt marsh		156		Wang and Cai[188]
9	Atlantic coast	salt marsh	185 ± 71	236 ± 120		Windham-Myers et al.[189]
10	Southern Moreton Bay, Australia	mangrove	259	927	285	Maher et al.[86]
11	Fukido River, Ishigaki Island, Japan	mangrove	193	857		Ohtsuka et al.[190]
12	North Brazilian	mangrove	22			Maher et al.[191]
13	Australian	mangrove	53			Maher et al.[191]
14	Sundarbans, India	mangrove	353	440	67	Ray et al.[192]
15	Florida, USA	mangrove	28		8	Romigh et al.[193]
16	African mangroves	mangrove	115		31	Machiwa[194]
17	Multiple mangrove creeks, Australia	mangrove		127		Sippo et al.[87]

to the loss of 30.8 Gg C $year^{-1}$ from C burial, with the total loss of 2.16 Tg C in the past 70 years. Fu et al.,[38] based on the Chinese and global decadal seagrass rates of decline, estimated the seagrass habitat loss rates in China at 533–3,100 ha $year^{-1}$ in the 1950s–2010s, equivalent to 11–282 Gg C $year^{-1}$ and 1.55–6.87 Tg C in total. To sum up, the sharp reduction of BCEs in China in the past 70 years has led to a great loss of the blue C function of these coastal wetlands, conserving the remaining BCEs in China can avoid 0.47–1.79 Tg C $year^{-1}$ emissions, which is also the most cost-effective pathway for achieving C neutrality.

However, the best way to conserve and restore BCEs wisely and ecologically is still largely under debate. For instance, the current BCE planting designs, aiming at promoting the wetland area have in some cases failed to sustain the growth of mangroves.[143] The replanting actions, such as mono-species plantation, decreased the diversity of the ecosystem and the possibility of resistance of pests and disease, and non-native mangrove species were widely used for mangrove restoration although they grow much faster than native species.[144,145] Therefore, the most desirable, comprehensive approaches need careful site selection and species selection, balancing ecological principles and economic benefits for mangrove restoration. For example, when choosing mangrove species for afforestation, it is recommended to prioritize local tree species. *Sonneratia apetala*, a common non-native mangrove species, had a high growth rate, but they are weak in biodiversity.[145] Native species *Bruguiera gymnorrhiza* mangroves have typical knee roots, and *Kandelia candel* forests have unique buttressing, which are good at accumulating the allochthonous C.[146] By estimating mangrove C storage, Liu et al.[16] indicated that C storage in mangroves could be increased by selecting specific species for afforestation and stand improvement practices, and the increased C stocks would be even more than the C gains by expanding the mangrove area. Chen et al.[147] proposed eco-farm systems to transfer aquaculture ponds to mangroves, suggesting a potentially broad application in southern China linking C sequestration and biomass gain in restored mangroves with required food production. Ouyang et al.[148] further demonstrated this approach by highlighting that the conversion of aquaculture ponds to mangroves could reduce CO_2 efflux. Other management options, which could substantially reduce CH_4 emissions from shrimp production, have also been introduced to enhance the dissolved oxygen availability in aquaculture.[149]

Regarding the large area of salt marsh and tidal flats in China, how to restore and increase the area of tidal marshes and flats will be of great importance for BCE restoration. Due to the relatively lower C sequestration of the unvegetated tidal flats compared with salt marshes,[150] transforming the unvegetated tidal flats to vegetated tidal marshes will also benefit C sequestration. However, there are still very few coastal wetland restoration projects considering these potentials. In addition, China's tidal flats are currently threatened by *Spartina alterniflora* invasion. In the past 30 years, nearly 467 km^2 of mudflats have been transformed into *S. alterniflora* salt marshes.[151] *S. alterniflora* invasion not only increases the input of plant biomass and organic litter[150] but also slows down the tidal flow, accelerates sediment deposition, and increases the sedimentation rate.[152,153] Generally, the total C sink of mudflats increases after *S. alterniflora* invasion, which favors the ecosystem C sequestration.[153,154] However, the invasion has led to changes of other ecosystem functions, such as biodiversity, so comprehensive ecosystem impactions after *S. alterniflora* invasion thus needs to be further evaluated.[66]

BCE management practices

Besides the conservation and restoration of BCEs, the management practices and technologies to improve C sequestration or reduce the C emissions of BCEs can also help to achieve the goals of C neutral. However, most studies on this topic have focused on terrestrial ecosystems,[140] such as forests and agriculture ecosystems, with very few reports on coastal wetlands. Here, we synthesize several technologies that have the potential to enhance the blue C sequestration in BCEs. However, some of these practices may also lead to the damage of these natural ecosystems, a balance between maximizing BCE C sequestration and the protection of these natural ecosystems is thus required during the implementation of these management practices.

Biochar addition. Biochar (BC) is a fine-grained and porous carbon-rich material made from a variety of organic materials (e.g., crop residues, woody materials, livestock manure, and other organic materials). BC is composed of alkyl structures and aromatic rings that form a core structure through close packing and high twisting. Due to its chemical and biological stability, BC can be stored in the ecosystem for a long time without being mineralized or decomposed,[155,156] which is of great significance for climate change mitigation. The incorporation of BC into soils can delay the return of C to the atmosphere as CO_2 or CH_4. Woolf et al.[157] roughly estimated the carbon sequestration potential of BC and found that BC could sequestrate 1.8 Pg CO_2-C per year (with no impact on food security and ecosystems), which is equivalent to 12% of annual anthropogenic GHG emissions. In addition, adding BC to soil can (1) increase soil pH for acidic soil remediation,[158] (2) increase soil cation exchange capacity and nutrient

Table 6. The reported values of ecological functions of coastal blue carbon ecosystems worldwide

			Total value (US$ ha^{-1} $year^{-1}$)		
Ecosystem function	Vegetation type	Study location	Low	High	Reference
Water conservation	whole wetland	Sinaloa, Mexico	1,906		Camacho-Valdez et al.[195]
	whole wetland	Catalan, Spain	1,287		Brenner et al.[196]
Storm/flood protection	whole wetland	China	9,678	255,000,000	Liu et al.[130]
	whole wetland	Catalan, Spain	68,166		Spalding et al.[124]
	whole wetland	Colorado, USA	40	20,830	Batker et al.[197]
	salt marshes	Galveston Island, USA	12,924		Feagin et al.[198]
	salt flat	Galveston Island, USA	405		Feagin et al.[198]
Water purification	whole wetland	British Columbia	1,395		Wilson et al.[199]
	whole wetland	Catalan, Spain	13,376		Brenner et al.[196]
	whole wetland	Thibodaux, USA	3,102	3,497	Ballard et al.[200]
	tidal marshes	global assessments	11,346		Purcell et al.[201]
Shoreline stabilization	whole wetland	Xiamen, China	3,400	10,100	Chen[132]
	whole wetland	USA	50	232	Ballard et al.[200]
Habitat provision/biodiversity	whole wetland	Qinghuangdao, China	350		Bai et al.[134]
	whole wetland	Catalan, Spain	497		Brenner et al.[196]
	estuaries	global assessments	222		Purcell et al.[201]
High productivity	salt marshes	Galveston Island, USA	6,944		Feagin et al.[198]
	tidal marshes	global assessments	790		Purcell et al.[201]
	estuaries	global assessments	882		Purcell et al.[201]

sequestration,[159] and (3) facilitate the adsorption and immobilization of heavy metals and organic pollutants.[160] Therefore, BC has multiple environmental effects, not only for C sequestration but also for reducing ocean pollution. Many studies have investigated the C sequestration potential of BC in farmlands[161] and forest ecosystems.[162] However, little is known about the C sequestration potential of BC in coastal wetland ecosystems.[163]

Gypsum addition. Gypsum ($CaSO_4 \cdot 2H_2O$) is typically used as a fertilizer to solve soil salinity problems. In the case of the coexistence of sulfate-reducing bacteria and methanogens in the environment, the addition of sulfate can inhibit the activity of methanogens, thereby inhibiting the release of methane.[164] At the same time, the reduction of sulfate leads to an increase in pH values. The dissolved metabolite CO_2 can react with Ca^{2+} to form carbonate precipitation, convert unstable organic carbon into inorganic carbon, and increase C sequestration.[165]

The role of gypsum in these system is mainly manifested in the following aspects: (1) gypsum is a slightly soluble mineral and plays the role of slow-release sulfate, (2) the continuous dissolution of gypsum converts the system into stable sulfate concentrations and enough electron acceptors, which promotes sulfate-reducing bacteria (SRB) to become the dominant microorganisms. SRB inhibits methanogens through competition for substrates and electrons, inhibiting the production of methane.[166] (3) Sulfate-reducing bacteria consume organic carbon while reducing sulfate, and accelerate the inorganic mineralization of OM.[167] The inorganic mineralization of OM produces carbonate and combines with Ca^{2+} in gypsum to form calcite, which improves the conversion rate of organic carbon to inorganic carbon. More experimental studies are needed to clarify the gypsum addition effects on the BCE C cyclings.

Iron oxide fertilization. Iron (Fe) is the most abundant redox-sensitive metal element in the Earth's crust, and iron oxides are widely present in wetland soils and sediments.[168] Fe(III) is the main electron acceptor for organic matter (OM) mineralization in the anaerobic environment of coastal wetlands. The process of dissimilatory Fe reduction can be considered as an important pathway for OM decomposition under anaerobic conditions.[168] Alternatively, iron oxides act as flocculants in the soil, forming organo-mineral complexes, and stabilizing soil aggregates for C sequestration.[169] The mechanism of organic C (OC) sequestration by iron oxides is mainly attributed to adsorption and co-precipitation. The exchange of ligands between hydroxyl and carboxyl groups in OM and the surface of iron oxides can adsorb up to 110–140 mg C g^{-1} by iron oxides.[170,171] For co-precipitation, organic carbon in soil has a high affinity for Fe^{3+}. The formation of occluded OM reduced the availability and increased the stability of OC.[172] At the same time, Fe(III) minerals can stimulate the growth of iron-reducing microorganisms. Through diverse metabolic pathways, microorganisms mineralize and utilize short-chain fatty acids, while forming more recalcitrant forms of OC.[173] Fe(II) oxidation also facilitates OC accumulation. For instance, a research from Yellowstone National Park found that at least 42% of DIC was incorporated to biomass C under Fe(II) oxidation by chemoautotrophs.[174] This suggest that, in some cases, chemoautotrophs can dominate the C fixation process.[174] In addition, Fe(III) reducers can inhibit methane production by competing with methanogenic archaea for acetate and hydrogen.[175] Thus, iron-reducing microorganisms could play a central role in SOM decomposition and GHG emissions from BCE soils, the fertilization of iron oxide may contribute the C sequestration in BCEs, and can be a potential technology for NbS of C neutrality.

Eutrophication. The eutrophication of near-shore waters is increasing due to land-based nitrogen entering offshore through surface runoff,[176] which is one of the most serious environmental problems facing the global coastal zone. Under tidal flux, large amounts of N typically enter coastal wetland ecosystems from runoff and can be retained by the coastal wetland vegetation. This changes the key processes of C cycle such as photosynthetic C input, plant-soil C partitioning, soil organic C decomposition, and soil-soluble organic C release.[177,178] Nitrogen is considered to be one of the most important nutrients limiting the net primary productivity of terrestrial ecosystems. Terrestrial N influences C sequestration in coastal wetlands through two main pathways: (1) most ecosystems are N limited, so exogenous N inputs can significantly increase the photosynthetic C sequestration capacity of plants by stimulating the growth of above- or below-ground plant parts, and thus increasing soil C inputs[177,179]; (2) N input affects soil organic C decomposition by altering soil N availability, regulating soil microbial growth, activity, community composition/diversity, and enzyme activity.[180] N input can change soil microbial biomass and microbial activity by increasing the

C and N content in the soil, leading to changes in the decomposition of SOM fractions. The increase of N content is more likely to result in reorganization of microorganisms and the formation of highly stable SOM.

CONCLUSION AND FUTURE PERSPECTIVES

Coastal wetlands are considered high-yielding habitats in sequestering significant amounts of atmospheric CO_2 globally. Understanding the magnitude and mechanisms of C cycling in the BCEs, such as salt marshes, mangroves, and seagrass, is at the foremost to resolve their role in coastal C budgets. Chinese coastal regions are dominated by salt marshes, with a much smaller area of mangrove forests and seagrass beds. Furthermore, the unvegetated tidal flat areas are large in this region. The conservation and restoration of these BCEs and improving their C sequestration potential by management can be a cost-effective pathway for climate change adaptation and help to achieve the C neutral goals of China in 2060. However, the current level of research on BCEs in China has not resolved the basic questions related to sequestration of these systems. Many problems still require scientific, socio-political, and technological solutions to enhance the C sequestration of Chinese BCEs for climate change mitigation. These include:

1. The area of these BCEs in China is still highly uncertain. The accuracy and spatiotemporal resolution of BCE distribution and GHG emissions need to be better resolved to monitor their C sequestration capacity. The potential of satellites and drones in monitoring BCE biomass should be fulfilled. Remote sensing approaches for monitoring seagrass meadows need new technological breakthroughs. Carrying out accurate C budget calculations of BCEs based on joint land-sea-sky observations is an important step toward science-based decision-making in China.[140]
2. Considering the large area of salt marshes and tidal flats in China, future field and modeling studies should prioritize these ecosystems to understand their carbon stock, burial rate, outwelling, and GHG emissions. Currently, the suspended sediment loads of major rivers are lower than the loads in the 20th century due to the construction of dams and soil erosion control measures.[59] Decreased sediment loads create an uncertain future for the C sequestration of these tidal ecosystems. Therefore, modeling and prediction of coastal ecosystem responses to human disturbance and climate change are also needed.
3. Even though the BCEs have a more effective C sequestration rate compared with terrestrial ecosystems, the Chinese coast has experienced extensive reclamation activities[142] and the construction of sea walls.[181] These modifications reduce the C stock and sequestration rates in BCEs as well as their ecosystem services such as landscape fragmentation, loss of biodiversity, destruction of habitats for fish and birds, the decline of fisheries resources, reduced water purification, increased water pollution, frequent harmful algal blooms, etc.[142] The impacts of these human disturbances on BCEs and their influence on ecosystem services need empirical research to guide future conservation and restoration. Moreover, policymakers should improve marine spatial planning, fully evaluate the negative impact of land reclamation and seawall construction, and enhance ocean awareness and public involvement in coastal management.
4. Restoration technologies including species selection and planting schemes, as well as management practices and technologies to improve planting success and enhance the C sequestration of BCEs are needed. Studies on this topic were mainly conducted in terrestrial ecosystems, with very rare reports on coastal wetlands. Technologies for CO_2 capture, utilization, and storage can also be applied in these coastal ecosystems, including fertilization with gypsum and BC.
5. Due to the high variation and quick response of BCEs to climate change and human disturbance, we suggest establishing a nation-wide coastal Blue Carbon Ecosystems Research Network. The network should combine long-term ecological studies, flux observations, remote sensing, and model simulations to quantify the C stock and flux of BCEs in China and their response to change. Only through long-term and real-time monitoring of C cycling in the BCEs can we clarify the impact of climate change and human activities on C sequestration of BCEs, and reveal the key drivers of the vertical C burial and lateral C exchange process. Overall, such a network would provide a reliable national Blue C inventory with uncertainties, and serve the national Carbon Neutral goals.

Considering the relatively high sedimentation rate of coastal wetlands in China, the total area of coastal wetlands should increase substantially by the end of the 21st century if there are no additional disturbances. The overall C sink and ecosystem services are expected to be further enhanced along with these efforts. However, the total area of coastal wetlands in China is relatively small in comparison with other terrestrial ecosystems due to past disturbance. Therefore, how to effectively restore and build BCEs, reduce the damage to the natural coastline around wetlands, improve their natural resilience, and enhance the ecosystem service of existing BCEs is of great importance to improve the "blue carbon" function of coastal wetlands in China. Overall, BCEs have a strong potential to become an NbS to mitigate climate change and support the Chinese C neutral goals in 2060.

REFERENCES

1. Miles, L., Agra, R., Sengupta, S., et al. (2021). Nature-based Solutions for Climate Change Mitigation (United Nations Environment Programme (UNEP), Nairobi and International Union for Conservation of Nature (IUCN), Gland).
2. Nellemann, C., Corcoran, E., Duarte, C., et al. (2009). Blue Carbon - the Role of Healthy Oceans in Binding Carbon (United Nations Environment Programme).
3. Jiao, N., Wang, H., Xu, G., et al. (2018). Blue carbon on the rise: challenges and opportunities. Natl. Sci. Rev. ***5***, 464–468.
4. Tang, J., Ye, S., Chen, X., et al. (2018). Coastal blue carbon: Concept, study method, and the application to ecological restoration. Sci. China Earth Sci. ***61***, 637–646.
5. Mcleod, E., Chmura, G.L., Bouillon, S., et al. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. Front. Ecol. Environ. ***9***, 552–560.
6. Macreadie, P.I., Costa, M.D.P., Atwood, T.B., et al. (2021). Blue carbon as a natural climate solution. Nat. Rev. Earth Environ. ***2***, 826–839.
7. Zinke, L. (2020). The colours of carbon. Nat. Rev. Earth Environ. ***1***, 141.
8. Spencer, T., Schuerch, M., Nicholls, R.J., et al. (2016). Global coastal wetland change under sea-level rise and related stresses: The DIVA Wetland Change Model. Global Planet. Change ***139***, 15–30.
9. Wang, F., Lu, X., Sanders, C.J., et al. (2019). Tidal wetland resilience to sea level rise increases their carbon sequestration capacity in United States. Nat. Commun. ***10***, 5434.
10. Wang, Z.A., Kroeger, K.D., Ganju, N.K., et al. (2016). Intertidal salt marshes as an important source of inorganic carbon to the coastal ocean. Limnol. Oceanogr. ***61***, 1916–1931.
11. Santos, I.R., Burdige, D.J., Jennerjahn, T.C., et al. (2021). The renaissance of Odum's outwelling hypothesis in 'Blue Carbon' science. Estuar. Coast Shelf Sci. ***255***, 107361.
12. Weston, N.B., Dixon, R.E., and Joye, S.B. (2006). Ramifications of increased salinity in tidal freshwater sediments: Geochemistry and microbial pathways of organic matter mineralization. J. Geophys. Res. ***111***, G01009.
13. Wang, F., Kroeger, K.D., Gonneea, M.E., et al. (2019). Water salinity and inundation control soil carbon decomposition during salt marsh restoration: An incubation experiment. Ecol. Evol. ***9***, 1911–1921.
14. Kroeger, K.D., Crooks, S., Moseman-Valtierra, S., et al. (2017). Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention. Sci. Rep. ***7***, 11914.
15. He, Y., Guan, W., Xue, D., et al. (2019). Comparison of methane emissions among invasive and native mangrove species in Dongzhaigang, Hainan Island. Sci. Total Environ. ***697***, 133945.
16. Liu, H., Ren, H., Hui, D., et al. (2014). Carbon stocks and potential carbon storage in the mangrove forests of China. J. Environ. Manag. ***133***, 86–93.
17. Murdiyarso, D., Purbopuspito, J., Kauffman, J.B., et al. (2015). The potential of Indonesian mangrove forests for global climate change mitigation. Nat. Clim. Change ***5***, 1089–1092.
18. Alongi, D.M. (2020). Carbon balance in salt marsh and mangrove ecosystems: A global synthesis. J. Mar. Sci. Eng. ***8***, 767.
19. Kauffman, J.B., Hughes, R.F., and Heider, C. (2009). Carbon pool and biomass dynamics associated with deforestation, land use, and agricultural abandonment in the neotropics. Ecol. Appl. ***19***, 1211–1222.
20. Khan, M.N.I., Suwa, R., and Hagihara, A. (2007). Carbon and nitrogen pools in a mangrove stand of *Kandelia obovata* (S., L.) Yong: vertical distribution in the soil–vegetation system. Wetl. Ecol. Manag. ***15***, 141–153.
21. Donato, D.C., Kauffman, J.B., Murdiyarso, D., et al. (2011). Mangroves among the most carbon-rich forests in the tropics. Nat. Geosci. ***4***, 293–297.
22. Kauffman, J.B., and Bhomia, R.K. (2017). Ecosystem carbon stocks of mangroves across broad environmental gradients in West-Central Africa: Global and regional comparisons. PLoS One ***12***, e0187749.
23. Hossain, M., and Nuruddin, A. (2016). Soil and mangrove: a review. J. Environ. Sci. Tech. ***9***, 198–207.
24. Kauffman, J.B., Adame, M.F., Arifanti, V.B., et al. (2020). Total ecosystem carbon stocks of mangroves across broad global environmental and physical gradients. Ecol. Monogr. ***90***, e01405.

25. Ouyang, X., and Lee, S.Y. (2020). Improved estimates on global carbon stock and carbon pools in tidal wetlands. Nat. Commun. ***11***, 317.
26. Jia, M., Wang, Z., Zhang, Y., et al. (2018). Monitoring loss and recovery of mangrove forests during 42 years: The achievements of mangrove conservation in China. Int. J. Appl. Earth Obs. ***73***, 535–545.
27. Mao, D., Wang, Z., Du, B., et al. (2020). National wetland mapping in China: A new product resulting from object-based and hierarchical classification of Landsat 8 OLI images. ISPRS J. Photogrammetry Remote Sens. ***164***, 11–25.
28. Herbeck, L.S., Krumme, U., Andersen, T.J., et al. (2020). Decadal trends in mangrove and pond aquaculture cover on Hainan (China) since 1966: mangrove loss, fragmentation and associated biogeochemical changes. Estuar. Coast Shelf Sci. ***233***, 106531.
29. Ma, C., Ai, B., Zhao, J., et al. (2019). Change Detection of Mangrove Forests in Coastal Guangdong during the Past Three Decades Based on Remote Sensing Data. Rem. Sens. ***11***, 921.
30. Wang, W., Fu, H., Lee, S.Y., et al. (2020). Can strict protection stop the decline of mangrove ecosystems in China? From rapid destruction to rampant degradation. Forests ***11***, 55.
31. Meng, Y., Bai, J., Gou, R., et al. (2021). Relationships between above-and below-ground carbon stocks in mangrove forests facilitate better estimation of total mangrove blue carbon. Carbon Bal. Manag. ***16***, 8–14.
32. Alongi, D.M. (2012). Carbon sequestration in mangrove forests. Carbon Manag. ***3***, 313–322.
33. Meng, W., Cui, W., Zhao, L., et al. (2019). The spatial distribution of blue carbon in the coastal wetlands of China. J. Biomed. Sci. ***26***, 13–20.
34. Twilley, R.R., Rovai, A.S., and Riul, P. (2018). Coastal morphology explains global blue carbon distributions. Front. Ecol. Environ. ***16***, 503–508.
35. Sanders, C.J., Maher, D.T., Tait, D.R., et al. (2016). Are global mangrove carbon stocks driven by rainfall? J. Geophys. Res. Biogeosci. ***121***, 2600–2609.
36. Mcowen, C., Weatherdon, L., Bochove, J.W., et al. (2017). A global map of saltmarshes. Biodivers. Data J. ***5***, e11764.
37. Hu, Y., Ni, M., Wen, Q., et al. (2021). Mapping coastal salt marshes in China using time series of Sentinel-1 SAR. Asia Pac. J. Clin. Nutr. ***30***, 122–129.
38. Fu, C., Li, Y., Zeng, L., et al. (2021). Stocks and losses of soil organic carbon from Chinese vegetated coastal habitats. Global Change Biol. ***27***, 202–214.
39. Duarte, C.M., Marba, N., Gacia, E., et al. (2010). Seagrass community metabolism: Assessing the carbon sink capacity of seagrass meadows. Global Biogeochem. Cycles ***24***, GB003793.
40. Short, F.T., and Green, E.P. (2004). World Atlas of Seagrasses. Bot. Mar. ***47***, 259–260.
41. McKenzie, L.J., Nordlund, L.M., Jones, B.L., et al. (2020). The global distribution of seagrass meadows. Environ. Res. Lett. ***15***, 074041.
42. Jayathilake, D.R., and Costello, M.J. (2018). A modelled global distribution of the seagrass biome. Biol. Conserv. ***226***, 120–126.
43. Fourqurean, J.W., Duarte, C.M., Kennedy, H., et al. (2012). Seagrass ecosystems as a globally significant carbon stock. Nat. Geosci. ***5***, 505–509.
44. McKenzie, L.J., Finkbeiner, M.A., and Kirkman, H. (2001). Methods for Mapping Seagrass Distribution (Global Seagrass Research Methods).
45. Orth, R.J., Carruthers, T.J.B., Dennison, W.C., et al. (2006). A global crisis for seagrass ecosystems. Bioscience ***56***, 987–996.
46. Zheng, F., Qiu, G., Fan, H., et al. (2013). Diversity, distribution and conservation of Chinese seagrass species. Biodivers. Sci. ***21***, 517–526.
47. Li, G., Wang, J., Hou, D., et al. (2016). Newly discovered seagrass beds and their ecological characteristics in Caofeidian, Bohai Sea. J. Environ. Sci. ***45***, 7–17.
48. Huang, X., Jiang, Z., Zhang, J., et al. (2010). Newly discovered seagrass beds of Guangdong Coast. Chin. J. Trop. Oceangraph. ***29***, 4–9.
49. Zhou, Y., Xu, S., Xu, S., et al. (2020). Newly discovered large-area seagrass beds (> 50ha) in temperate seas of China: iii population dynamics and replenishment mechanism of eelgrass in xingcheng-juehua island area of bohai sea. Oceans and Lakes ***51***, 943–951.
50. Zhou, Y., Xu, S., Xu, S., et al. (2019). The largest seagrass bed of Eelgrass in China was discovered in the coastal waters of Tangshan, Bohai Sea. Mar. Sci. ***43***, 50–55.
51. Li, J., Yang, D., Zhao, D., et al. (2019). Analysis of blue carbon in China's coastal zone. Seizure ***71***, 207–213.
52. Howard, J., Hoyt, S., and Isensee, K. (2014). Coastal Blue Carbon: Methods for Assessing Carbon Stocks and Emissions Factors in Mangroves, Tidal Salt Marshes, and Seagrasses (Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature.).
53. Gao, Y.P. (2013). Seagrass meadow carbon sink and amplification of the carbon sink for eelgrass bed in Sanggou Bay. Prog. Fish. Sci. ***34***, 17–21.
54. Huang, Y.H., Lee, C.L., Chung, C.Y., et al. (2015). Carbon budgets of multispecies seagrass beds at Dongsha Island in the South China Sea. Mar. Environ. Res. ***106***, 92–102.
55. Li, M. (2018). Carbon Storage of Seagrass Bed Sediments in Guangxi (Guangxi Normal University).
56. Jiang, Z., Liu, S., Zhang, J., et al. (2017). Newly discovered seagrass beds and their potential for blue carbon in the coastal seas of Hainan Island, South China Sea. Mar. Pollut. Bull. ***125***, 513–521.
57. Murray, N.J., Phinn, S.R., DeWitt, M., et al. (2019). The global distribution and trajectory of tidal flats. Nature ***565***, 222–225.
58. Chen, Z.L., and Lee, S.Y. (2022). Tidal Flats as a Significant Carbon Reservoir in Global Coastal Ecosystems. Front. Mar. Sci. ***9***, 900896.
59. Chen, J., Wang, D., Li, Y., et al. (2020). The carbon stock and sequestration rate in tidal flats from coastal China. Global Biogeochem. Cycles ***34***. e2020gb006772.
60. Twilley, R.R., Chen, R.H., and Hargis, T. (1992). Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. Water Air Soil Pollut. ***64***, 265–288.
61. Breithaupt, J.L., Smoak, J.M., Smith, T.J., et al. (2012). Organic carbon burial rates in mangrove sediments: Strengthening the global budget. Global Biogeochem. Cycles ***26***. 2012GB004375.
62. Rosentreter, J.A., Maher, D.T., Erler, D.V., et al. (2018). Methane emissions partially offset "blue carbon" burial in mangroves. Sci. Adv. ***4***, eaao4985.
63. Bouillon, S., Borges, A.V., Castaneda-Moya, E., et al. (2008). Mangrove production and carbon sinks: A revision of global budget estimates. Global Biogeochem. Cycles ***22***, GB2013.
64. Giri, C., Ochieng, E., Tieszen, L.L., et al. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data. Global Ecol. Biogeogr. ***20***, 154–159.
65. Wang, F., Sanders, C.J., Santos, I.R., et al. (2021). Global blue carbon accumulation in tidal wetlands increases with climate change. Natl. Sci. Rev. ***8***, nwaa296.
66. Wang, F., Tang, J., Ye, S., et al. (2021). The blue carbon function of Chinese coastal wetlands and countermeasures to perform carbon neutral strategy. Bull. Chin. Acad. Sci. ***36***, 241–251.
67. Ren, H., Chen, H., Li, Z., et al. (2010). Biomass accumulation and carbon storage of four different aged *Sonneratia apetala* plantations in Southern China. Plant Soil ***327***, 279–291.
68. Alongi, D.M. (2018). Blue Carbon:Coastal Sequestration for Climate Change Mitigation (Springer Nature).
69. Scholz, M. (2010). Wetland Systems: Storm Water Management Control (Springer Science & Business Media).
70. Duarte, C.M., Losada, I.J., Hendriks, I.E., et al. (2013). The role of coastal plant communities for climate change mitigation and adaptation. Nat. Clim. Change ***3***, 961–968.
71. Duarte, C.M., Middelburg, J.J., and Caraco, N. (2005). Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences ***2***, 1–8.
72. Pidgeon, E. (2009). Carbon sequestration by coastal marine habitats: Important missing sinks. In The Management of Natural Coastal Carbon Sinks, D.d.A. Laȭoley and G. Grimsditch, eds.
73. Kennedy, H., Beggins, J., Duarte, C.M., et al. (2010). Seagrass sediments as a global carbon sink: Isotopic constraints. Global Biogeochem. Cycles ***24***, GB003848.
74. Mazarrasa, I., Marbà, N., Lovelock, C.E., et al. (2015). Seagrass meadows as a globally significant carbonate reservoir. Biogeosciences ***12***, 4993–5003.
75. Saderne, V., Geraldi, N.R., Macreadie, P.I., et al. (2019). Role of carbonate burial in Blue Carbon budgets. Nat. Commun. ***10***, 1106.
76. Hernández, M.E., and Junca-Gómez, D. (2020). Carbon stocks and greenhouse gas emissions in mangroves with different vegetation assemblies in the central coastal plain of Veracruz Mexico. Sci. Total Environ. ***741***, 140276.
77. Sasmito, S.D., Taillardat, P., Clendenning, J.N., et al. (2019). Effect of land-use and land-cover change on mangrove blue carbon: A systematic review. Global Change Biol. ***25***, 4291–4302.
78. Chen, G.C., Tam, N.F.Y., and Ye, Y. (2010). Summer fluxes of atmospheric greenhouse gases N_2O, CH_4 and CO_2 from mangrove soil in South China. Sci. Total Environ. ***408***, 2761–2767.
79. Rosentreter, J.A., Maher, D.T., Erler, D.V., et al. (2018). Factors controlling seasonal CO_2 and CH_4 emissions in three tropical mangrove-dominated estuaries in Australia. Estuar. Coast Shelf Sci. ***215***, 69–82.
80. Ma, J., Ullah, S., Niu, A., et al. (2021). Heavy metal pollution increases CH_4 and decreases CO_2 emissions due to soil microbial changes in a mangrove wetland: Microcosm experiment and field examination. Chemosphere ***269***, 128735.
81. Chen, G.C., Ulumuddin, Y.I., Pramudji, S., et al. (2014). Rich soil carbon and nitrogen but low atmospheric greenhouse gas fluxes from North Sulawesi mangrove swamps in Indonesia. Sci. Total Environ. ***487***, 91–96.
82. Jeffrey, L.C., Reithmaier, G., Sippo, J.Z., et al. (2019). Are methane emissions from mangrove stems a cryptic carbon loss pathway? Insights from a catastrophic forest mortality. New Phytol. ***224***, 146–154.
83. Chen, Y., Chen, G., and Ye, Y. (2015). Coastal vegetation invasion increases greenhouse gas emission from wetland soils but also increases soil carbon accumulation. Sci. Total Environ. ***526***, 19–28.
84. Yuan, J., Ding, W., Liu, D., et al. (2015). Exotic Spartina alterniflora invasion alters ecosystem-atmosphere exchange of CH_4 and N_2O and carbon sequestration in a coastal salt marsh in China. Global Change Biol. ***21***, 1567–1580.
85. Li, X., Sardans, J., Hou, L., et al. (2020). Climatic temperature controls the geographical patterns of coastal marshes greenhouse gases emissions over China. J. Hydrol. X. ***590***, 125378.
86. Maher, D.T., Call, M., Santos, I.R., et al. (2018). Beyond burial: lateral exchange is a significant atmospheric carbon sink in mangrove forests. Biol. Lett. ***14***, 20180200.
87. Sippo, J.Z., Maher, D.T., Tait, D.R., et al. (2016). Are mangroves drivers or buffers of coastal acidification? Insights from alkalinity and dissolved inorganic carbon export estimates across a latitudinal transect. Global Biogeochem. Cycles ***30***, 753–766.
88. Jiao, N., Herndl, G.J., Hansell, D.A., et al. (2011). The microbial carbon pump and the oceanic recalcitrant dissolved organic matter pool. Nat. Rev. Microbiol. ***9***, 555.
89. Cabral, A., Dittmar, T., Call, M., et al. (2021). Carbon and alkalinity outwelling across the groundwater-creek-shelf continuum off Amazonian mangroves. Limnol. Oceanogr. Lett. ***6***, 369–378.
90. Jiao, N., Liu, J., Jiao, F., et al. (2020). Microbes mediated comprehensive carbon sequestration for negative emissions in the ocean. Natl. Sci. Rev. ***7***, 1858–1860.

91. Blankespoor, B., Dasgupta, S., and Laplante, B. (2014). Sea-level rise and coastal wetlands. Ambio **43**, 996–1005.
92. Simard, M., Fatoyinbo, L., Smetanka, C., et al. (2018). Mangrove canopy height globally related to precipitation, temperature and cyclone frequency. Nat. Geosci. **12**, 40–45.
93. Spivak, A.C., Sanderman, J., Bowen, J.L., et al. (2019). Global-change controls on soil-carbon accumulation and loss in coastal vegetated ecosystems. Nat. Geosci. **12**, 685–692.
94. Alongi, D.M. (2009). The Energetics of Mangrove Forests (Springer Netherlands).
95. Leopold, A., Marchand, C., Renchon, A., et al. (2016). Net ecosystem CO_2 exchange in the "Coeur de Voh" mangrove, New Caledonia: Effects of water stress on mangrove productivity in a semi-arid climate. Agric. For. Meteorol. **223**, 217–232.
96. Liu, J., and Lai, D.Y. (2019). Subtropical mangrove wetland is a stronger carbon dioxide sink in the dry than wet seasons. Agric. For. Meteorol. **278**, 107644.
97. Turner, R.E. (1976). Geographic variations in salt marsh macrophyte production: A review. Contrib. Mar. Sci. **20**, 47–66.
98. Ward, R.D., Friess, D.A., Day, R.H., et al. (2016). Impacts of climate change on mangrove ecosystems: a region by region overview. Ecosys. Health Sustain. **2**, e01211.
99. Morris, J., Sundberg, K., and Hopkinson, C. (2013). Salt Marsh Primary Production and Its Responses to Relative Sea Level and Nutrients in Estuaries at Plum Island, Massachusetts, and North Inlet, South Carolina, USA. Oceanography **26**, 78–84.
100. Ouyang, X., Kristensen, E., Zimmer, M., et al. (2023). Response of macrophyte litter decomposition in global blue carbon ecosystems to climate change. Global Change Biol. **29**, 3806–3820.
101. Alongi, D.M. (2008). Mangrove forests: Resilience, protection from tsunamis, and responses to global climate change. Estuar. Coast Shelf Sci. **76**, 1–13.
102. Schuerch, M., Spencer, T., Temmerman, S., et al. (2018). Future response of global coastal wetlands to sea-level rise. Nature **561**, 231–234.
103. Krauss, K.W., McKee, K.L., Lovelock, C.E., et al. (2014). How mangrove forests adjust to rising sea level. New Phytol. **202**, 19–34.
104. Woodroffe, C.D., Rogers, K., McKee, K.L., et al. (2016). Mangrove Sedimentation and Response to Relative Sea-Level Rise. Ann. Rev. Mar. Sci. **8**, 243–266.
105. Vahsen, M.L., Blum, M.J., Megonigal, J.P., et al. (2023). Rapid plant trait evolution can alter coastal wetland resilience to sea level rise. Science **379**, 393–398.
106. Kristensen, E., Connolly, R., Ferreira, T.O., et al. (2017). Biogeochemical cycles; global approaches and perspectives. In Mangrove Ecosystems: A Global Biogeographic Perspective Structure, Function and Services (Springer).
107. Ouyang, X., Lee, S.Y., and Connolly, R.M. (2017). The role of root decomposition in global mangrove and saltmarsh carbon budgets. Earth Sci. Rev. **166**, 53–63.
108. Lewis, D.B., Brown, J.A., and Jimenez, K.L. (2014). Effects of flooding and warming on soil organic matter mineralization in Avicennia germinans mangrove forests and Juncus roemerianus salt marshes. Estuar. Coast Shelf Sci. **139**, 11–19.
109. Yin, S., An, S., Deng, Q., et al. (2015). Spartina alterniflora invasions impact CH_4 and N_2O fluxes from a salt marsh in eastern China. Ecol. Eng. **81**, 192–199.
110. Chambers, L.G., Davis, S.E., Troxler, T., et al. (2014). Biogeochemical effects of simulated sea level rise on carbon loss in an Everglades mangrove peat soil. Hydrobiologia **726**, 195–211.
111. Davidson, E.A., and Janssens, I.A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature **440**, 165–173.
112. Janssens, I.A., Freibauer, A., Ciais, P., et al. (2003). Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic CO2 emissions. Science **300**, 1538–1542.
113. Jacotot, A., Marchand, C., and Allenbach, M. (2019). Biofilm and temperature controls on greenhouse gas CO_2 and CH_4 emissions from a Rhizophora mangrove soil (New Caledonia). Sci. Total Environ. **650**, 1019–1028.
114. Kirwan, M.L., and Megonigal, J.P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. Nature **504**, 53–60.
115. Pendleton, L., Donato, D.C., Murray, B.C., et al. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One **7**, e43542.
116. Gedan, K.B., Silliman, B.R., and Bertness, M.D. (2009). Centuries of human-driven change in salt marsh ecosystems. Ann. Rev. Mar. Sci **1**, 117–141.
117. Coleman, J.M., Huh, O.K., and Braud, D. (2008). Wetland loss in world deltas. J. Coast Res. **1**, 1–14.
118. Waycott, M., Duarte, C.M., Carruthers, T.J.B., et al. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. USA **106**, 12377–12381.
119. de Los Santos, C.B., Krause-Jensen, D., Alcoverro, T., et al. (2019). Recent trend reversal for declining European seagrass meadows. Nat. Commun. **10**, 3356.
120. Hamilton, S.E., and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). Global Ecol. Biogeogr. **25**, 729–738.
121. Sanderman, J., Hengl, T., Fiske, G., et al. (2018). A global map of mangrove forest soil carbon at 30 m spatial resolution. Environ. Res. Lett. **13**, 055002.
122. Millennium Ecosystem Assessment, M. (2005). Ecosystems and Human Well-Being: Current State and Trends (Island Press), pp. 663–679.
123. Barbier, E.B., Hacker, S.D., Kennedy, C., et al. (2011). The value of estuarine and coastal ecosystem services. Ecol. Monogr. **81**, 169–193.
124. Spalding, M.D., Ruffo, S., Lacambra, C., et al. (2014). The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. Ocean Coast Manag. **90**, 50–57.
125. Costanza, R., de Groot, R., Sutton, P., et al. (2014). Changes in the global value of ecosystem services. Global Environ. Change **26**, 152–158.
126. De Groot, R., Brander, L., Van Der Ploeg, S., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. Ecosyst. Serv. **1**, 50–61.
127. Sun, X., Li, Y., Zhu, X., et al. (2017). Integrative assessment and management implications on ecosystem services loss of coastal wetlands due to reclamation. J. Clean. Prod. **163**, S101–S112.
128. Costanza, R., Pérez-Maqueo, O., Martinez, M.L., et al. (2008). The value of coastal wetlands for hurricane protection. Ambio **37**, 241–248.
129. Simpson, R.H., and Riehl, H. (1981). The Hurricane and its Impact (Louisiana State University Press).
130. Liu, X., Wang, Y., Costanza, R., et al. (2019). The value of China's coastal wetlands and seawalls for storm protection. Ecosyst. Serv. **36**, 100905.
131. Breaux, A., Farber, S., and Day, J. (1995). Using natural coastal wetlands systems for wastewater treatment: an economic benefit analysis. J. Environ. Manag. **44**, 285–291.
132. Chen, P. (2006). Evaluation on service value of wetland ecosystem in Xiamen City. Wetl. Sci. **4**, 101–106.
133. Xie, G., Zhang, C., Zhang, C., et al. (2015). The value of ecosystem services in China. Resour. Sci. **37**, 1740–1746.
134. Bai, X., Hao, Y., Zhao, Z., et al. (2019). Valuation of ecosystem services of coastal wetlands in Qinhuangdao, China. Aquat. Ecosys. Health Manag. **22**, 205–214.
135. Worthington, T., and Spalding, M. (2018). Mangrove Restoration Potential: A global map highlighting a critical opportunity. Tech. Rep.
136. Su, J., Friess, D.A., and Gasparatos, A. (2021). A meta-analysis of the ecological and economic outcomes of mangrove restoration. Nat. Commun. **12**, 5050.
137. Wang, F., Eagle, M., Kroeger, K.D., et al. (2021). Plant biomass and rates of carbon dioxide uptake are enhanced by successful restoration of tidal connectivity in salt marshes. Sci. Total Environ. **750**, 141566.
138. Griscom, B.W., Adams, J., Ellis, P.W., et al. (2017). Natural climate solutions. Proc. Natl. Acad. Sci. USA **114**, 11645–11650.
139. Bossio, D.A., Cook-Patton, S.C., Ellis, P.W., et al. (2020). The role of soil carbon in natural climate solutions. Nat. Sustain. **3**, 391–398.
140. Wang, F., Harindintwali, J.D., Yuan, Z., et al. (2021). Technologies and perspectives for achieving carbon neutrality. Innovation **2**, 100180.
141. An, S., Li, H., Guan, B., et al. (2007). China's natural wetlands: past problems, current status, and future challenges. Ambio **36**, 335–342.
142. Wang, W., Liu, H., Li, Y., et al. (2014). Development and management of land reclamation in China. Ocean Coast Manag. **102**, 415–425.
143. Lee, S.Y., Hamilton, S., Barbier, E.B., et al. (2019). Better restoration policies are needed to conserve mangrove ecosystems. Nat. Ecol. Evol. **3**, 870–872.
144. He, Z., Peng, Y., Guan, D., et al. (2018). Appearance can be deceptive: shrubby native mangrove species contributes more to soil carbon sequestration than fast-growing exotic species. Plant Soil **432**, 425–436.
145. Ren, H., Lu, H., Shen, W., et al. (2009). *Sonneratia apetala* Buch.Ham in the mangrove ecosystems of China: An invasive species or restoration species? Ecol. Eng. **35**, 1243–1248.
146. Qiu, L. (2019). The Influence of aerial root/belowground root structure on the vertical accretion and elevation change of mangroves (Xiamen university). Master Degree.
147. Chen, L., Fan, H., Su, Z., et al. (2021). Enhancing carbon storage in mangrove ecosystems of China through sustainable restoration and aquaculture actions. In Wetland Carbon and Environmental Management, pp. 127–141.
148. Ouyang, X., Lee, S., and Wang, W. (2021). The 'Perfect' Conversion: Dramatic Increase in CO_2 Efflux from Shellfish Ponds and Mangrove Conversion in China. Sustainability **13**, 13163.
149. Lai, D.Y., Yang, P., and Tong, C. (2021). The impacts of aquaculture activities on greenhouse gas dynamics in the subtropical estuarine zones of China. In Wetland Carbon and Environmental Management, pp. 213–227.
150. Lin, W.J., Wu, J., and Lin, H.J. (2020). Contribution of unvegetated tidal flats to coastal carbon flux. Global Change Biol. **26**, 3443–3454.
151. Mao, D., Liu, M., Wang, Z., et al. (2019). Rapid invasion of *Spartina slterniflora* in the coastal zone of mainland China: spatiotemporal patterns and human prevention. Sensors **19**, 2308.
152. Huang, M., Ge, C., Zuo, P., et al. (2018). The contribution of Spartina introduction on organic matter source and its effects on carbon burial in tidal flats. J. Nanjing Univ. (Nat. Sci.) **54**, 655–664.
153. Dan, W., Rong, Z., Jun, X., et al. (2015). Contribution of invasive species Spartina alterniflora to soil organic carbon pool in coastal wetland: Stable isotope approach. Chin. J. Plant Ecol. **39**, 941–949.
154. Xie, X.F., Sun, X.M., Wu, T., et al. (2020). Impacts of *Spartina alterniflora* invasion on coastal wetland ecosystem: advances and prospects. Chin. J. Appl. Ecol. **31**, 2119–2128.
155. Calvelo Pereira, R., Kaal, J., Camps Arbestain, M., et al. (2011). Contribution to characterisation of biochar to estimate the labile fraction of carbon. Org. Geochem. **42**, 1331–1342.
156. Singh, B.P., Cowie, A.L., and Smernik, R.J. (2012). Biochar carbon stability in a clayey soil as a function of feedstock and pyrolysis temperature. Environ. Sci. Technol. **46**, 11770–11778.
157. Woolf, D., Amonette, J.E., Street-Perrott, F.A., et al. (2010). Sustainable biochar to mitigate global climate change. Nature **1**, 56–59.
158. Zhang, Y., Wang, J., and Feng, Y. (2021). The effects of biochar addition on soil physicochemical properties: A review. Catena **202**, 105284.
159. Hossain, M.Z., Bahar, M.M., Sarkar, B., et al. (2020). Biochar and its importance on nutrient dynamics in soil and plant. Biochar **2**, 379–420.

160. Xu, P., Sun, C.X., Ye, X.Z., et al. (2016). The effect of biochar and crop straws on heavy metal bioavailability and plant accumulation in a Cd and Pb polluted soil. Ecotoxicol. Environ. Saf. ***132***, 94–100.
161. Liu, S., Zhang, Y., Zong, Y., et al. (2016). Response of soil carbon dioxide fluxes, soil organic carbon and microbial biomass carbon to biochar amendment: a meta-analysis. GCB Bioenergy ***8***, 392–406.
162. Mitchell, P.J., Simpson, A.J., Soong, R., et al. (2015). Shifts in microbial community and water-extractable organic matter composition with biochar amendment in a temperate forest soil. Soil Biol. Biochem. ***81***, 244–254.
163. Luo, X., Wang, L., Liu, G., et al. (2016). Effects of biochar on carbon mineralization of coastal wetland soils in the Yellow River Delta, China. Ecol. Eng. ***94***, 329–336.
164. Theint, E.E., Bellingrath-Kimura, S.D., Oo, A.Z., et al. (2016). Influence of Gypsum Amendment on Methane Emission from Paddy Soil Affected by Saline Irrigation Water. Front. Environ. Sci. ***3***, 79.
165. Zhang, H., Liu, R., and Lal, R. (2016). Optimal sequestration of carbon dioxide and phosphorus in soils by gypsum amendment. Environ. Chem. Lett. ***14***, 443–448.
166. St James, A.R., and Richardson, R.E. (2021). Stimulation of dissimilatory sulfate reduction in response to sulfate in microcosm incubations from two contrasting temperate peatlands near Ithaca, NY, USA. FEMS Microbiol. Lett. ***368***, fnab153.
167. Deng, Q., Wu, X., Wang, Y., et al. (2018). Activity characteristics of sulfate reducing bacteria and formation mechanism of hydrogen sulfide. Appl. Ecol. Environ. Res. ***16***, 6369–6383.
168. Liu, Y., Dong, Y., Ge, T., et al. (2019). Impact of prolonged rice cultivation on coupling relationship among C, Fe, and Fe-reducing bacteria over a 1000-year paddy soil chronosequence. Biol. Fertil. Soils ***55***, 589–602.
169. Duan, X., Yu, X., Li, Z., et al. (2020). Iron-bound organic carbon is conserved in the rhizosphere soil of freshwater wetlands. Soil Biol. Biochem. ***149***, 107949.
170. Chen, C., Dynes, J.J., Wang, J., et al. (2014). Properties of Fe-organic matter associations via coprecipitation versus adsorption. Environ. Sci. Technol. ***48***, 13751–13759.
171. Zhao, Q., Adhikari, D., Huang, R., et al. (2016). Coupled dynamics of iron and iron-bound organic carbon in forest soils during anaerobic reduction. Chem. Geol.
172. Wagai, R., and Mayer, L.M. (2007). Sorptive stabilization of organic matter in soils by hydrous iron oxides. Geochem. Cosmochim. Acta ***71***, 25–35.
173. Xu, J.X., Li, X.M., Sun, G.X., et al. (2019). Fate of labile organic carbon in paddy soil is regulated by microbial ferric iron reduction. Environ. Sci. Technol. ***53***, 8533–8542.
174. Jennings, R.M., Whitmore, L.M., Moran, J.J., et al. (2014). Carbon dioxide fixation by Metallosphaera yellowstonensis and acidothermophilic iron-oxidizing microbial communities from Yellowstone National Park. Appl. Environ. Microbiol. ***80***, 2665–2671.
175. Fan, L., Dippold, M.A., Ge, T., et al. (2020). Anaerobic oxidation of methane in paddy soil: Role of electron acceptors and fertilization in mitigating CH4 fluxes. Soil Biol. Biochem. ***141***, 107685.
176. Deegan, L.A., Johnson, D.S., Warren, R.S., et al. (2012). Coastal eutrophication as a driver of salt marsh loss. Nature ***490***, 388–392.
177. Herbert, E.R., Schubauer-Berigan, J.P., and Craft, C.B. (2020). Effects of 10 yr of nitrogen and phosphorus fertilization on carbon and nutrient cycling in a tidal freshwater marsh. Limnol. Oceanogr. ***65***, 1–19.
178. Hu, Y., Wang, L., Fu, X., et al. (2016). Salinity and nutrient contents of tidal water affects soil respiration and carbon sequestration of high and low tidal flats of Jiuduansha wetlands in different ways. Sci. Total Environ. ***565***, 637–648.
179. Fernández-Martínez, M., Vicca, S., Janssens, I.A., et al. (2014). Nutrient availability as the key regulator of global forest carbon balance. Nat. Clim. Change ***4***, 471–476.
180. Qu, W., Han, G., Eller, F., et al. (2020). Nitrogen input in different chemical forms and levels stimulates soil organic carbon decomposition in a coastal wetland. Catena ***194***, 104672.
181. Ma, Z., Melville, D.S., Liu, J., et al. (2014). Rethinking China's new great wall. Science ***346***, 912–914.
182. Ganju, N.K., Defne, Z., Elsey-Quirk, T., et al. (2019). Role of tidal wetland stability in lateral fluxes of particulate organic matter and carbon. J. Geophys. Res. Biogeosci. ***124***, 1265–1277.
183. Bogard, M.J., Bergamaschi, B.A., Butman, D.E., et al. (2020). Hydrologic export is a major component of coastal wetland carbon budgets. Global Biogeochem. Cycles ***34***, GB006430.
184. Correa, R.E., Xiao, K., Conrad, S.R., et al. (2021). Groundwater carbon exports exceed sediment carbon burial in a salt marsh. Estuar. Coast ***45***, 1545–1561.
185. Czapla, K.M., Anderson, I.C., and Currin, C.A. (2020). Net ecosystem carbon balance in a North Carolina, USA, salt marsh. J. Geophys. Res. Biogeosci. ***125***. e2019JG005509.
186. Guimond, J., and Tamborski, J. (2021). Salt marsh hydrogeology: a review. Water ***13***, 543.
187. Neubauer, S.C., and Anderson, I.C. (2003). Transport of dissolved inorganic carbon from a tidal freshwater marsh to the York River estuary. Limnol. Oceanogr. ***48***, 299–307.
188. Wang, Z.A., and Cai, W.J. (2004). Carbon dioxide degassing and inorganic carbon export from a marsh-dominated estuary (the Duplin River): A marsh CO2 pump. Limnol. Oceanogr. ***49***, 341–354.
189. Windham-Myers, L., Cai, W.-J., Alin, S.R., et al. (2018). Chapter 15: Tidal wetlands and estuaries. In Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report, N. Cavallaro, G. Shrestha, and R. Birdsey, et al., eds.
190. Ohtsuka, T., Onishi, T., Yoshitake, S., et al. (2020). Lateral export of dissolved inorganic and organic carbon from a small mangrove estuary with tidal fluctuation. Forests ***11***, 1041.
191. Maher, D.T., Santos, I.R., Golsby-Smith, L., et al. (2013). Groundwater-derived dissolved inorganic and organic carbon exports from a mangrove tidal creek: The missing mangrove carbon sink? Limnol. Oceanogr. ***58***, 475–488.
192. Ray, R., Baum, A., Rixen, T., et al. (2018). Exportation of dissolved (inorganic and organic) and particulate carbon from mangroves and its implication to the carbon budget in the Indian Sundarbans. Sci. Total Environ. ***621***, 535–547.
193. Romigh, M.M., Davis, S.E., Rivera-Monroy, V.H., et al. (2006). Flux of organic carbon in a riverine mangrove wetland in the Florida Coastal Everglades. Hydrobiologia ***569***, 505–516.
194. Machiwa, J.F. (1999). Lateral fluxes of organic carbon in a mangrove forest partly contaminated with sewage wastes. Mangroves Salt Marshes ***3***, 95–104.
195. Camacho-Valdez, V., Ruiz-Luna, A., Ghermandi, A., et al. (2013). Valuation of ecosystem services provided by coastal wetlands in northwest Mexico. Ocean Coast Manag. ***78***, 1–11.
196. Brenner, J., Jiménez, J.A., Sardá, R., et al. (2010). An assessment of the non-market value of the ecosystem services provided by the Catalan coastal zone, Spain. Ocean Coast Manag. ***53***, 27–38.
197. Batker, D., Christin, Z., Graf, W., et al. (2014). Nature's value in the Colorado River Basin. Earth Economics.
198. Feagin, R.A., Martinez, M.L., Mendoza-Gonzalez, G., et al. (2010). Salt marsh zonal migration and ecosystem service change in response to global sea level rise: a case study from an urban region. Ecol. Soc. ***15***, art14.
199. Wilson, S.J., Nutt, D.J., Alford, C., et al. (2010). British Association for Psychopharmacology consensus statement on evidence-based treatment of insomnia, parasomnias and circadian rhythm disorders. J. Psychopharm. ***24***, 1577–1601.
200. Ballard, J., Pezda, J., and Spencer, D. (2016). An Economic Valuation of Southern California Coastal Wetlands (Bren School of Environmental Science and Management).
201. Purcell, A., Khanal, P., Straka, T., et al. (2020). Valuing Ecosystem Services of Coastal Marshes and Wetlands (Land-Grant Press).

ACKNOWLEDGMENTS

This paper is in memory of Prof. Jianwu Tang (1970–2023) for his pioneering contributions to blue carbon research. His groundbreaking work expanded our scientific understanding of coastal blue carbon, inspired many researchers to work toward mitigating the impacts of climate change, and his unwavering support for young scientists will be his enduring legacy. He will be greatly missed, and his influence will continue to inspire us to explore new frontiers in ecological research. This work is funded by the CAS Project for Young Scientists in Basic Research (YSBR-037), ANSO collaborative research (ANSO-CR-KP-2022-11), the National Natural Science Foundation of China (U2106209, 42141003, 42141016, and 32171594), the Guangdong Basic and Applied Basic Research Foundation (2021B1515020011 and 2021B1212110004), the CAS Youth Innovation Promotion Association (2021347), the National Forestry and Grassland Administration Youth Talent Support Program (2020BJ003), and the R&D program of Guangdong Provincial Department of Science and Technology (2018B030324003).

AUTHOR CONTRIBUTIONS

H.R. and N.J. provided direction and guidance throughout the preparation of this article. F.W. collected and interpreted studies and was a major contributor to the writing and editing of the manuscript. All authors revised and approved the final manuscript.

DECLARATION OF INTERESTS

The authors declare no competing interests.

SUPPLEMENTAL INFORMATION

It can be found online at https://doi.org/10.1016/j.xinn.2023.100481.

LEAD CONTACT WEBSITE

https://www.scholarmate.com/P/famingw