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New insight into global blue carbon estimation under human activity in land-sea interaction area: A case study of China

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ABSTRACT

The C sequestration in coastal blue carbon (C_b) ecosystems, including mangroves, seagrasses and saltmarshes, was discovered to be useful in mitigating the increasing trend of carbon dioxide (CO_2) emission due to climate change. In this study, we systematically estimate traditional C_b ecosystem distribution and the associated C_b sequestration rate, and then further quantify the C_b sinks fishery contribution to C_b ecosystem due to human activity in coastal ecosystem. The results show that the global C_b ecosystem is able to store 10.8 PgC, wherein biomass and soil are able to store 2.13 and 8.68 PgC, respectively. In China, the C_b pools are 162 TgC in mangroves, 67 TgC in saltmarshes and 75 TgC in seagrass. The human activity induced global C_b sink fishery on C_b ecosystem is about 26.58–37.6 TgC yr^{-1}, accounting for 30.7%–43.4% of the world's traditional C_b sequestration ecosystem.The global C_b sequestration potential reaches up to 86.59 Tg yr^{-1}, while China can explain 1.70% of the world's total C_b sequestration. However, in China, the C_b sequestration due to human activity reaches up to 6.32–7.89 TgC yr^{-1}, accounting for 20.9%–23.7% of global C_b sink fishery. Therefore, it is very important to build the C_b sink fisheries measure and monitor system to scientifically valuate C_b sink fisheries and associated development potential.

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1. Introduction

Traditionally, the blue carbon (C_b) is defined as the C captured by living organisms in oceans that stored in the form of sediments from mangroves, saltmarshes and seagrasses (Siikamäki et al., 2012), The costal vegetated habitats, in particular mangroves, saltmarshes and seagrasses only cover less than 2% of the area of the world's oceans, but sequester at least 50% of the C stored in ocean sediments (Nellemann et al., 2009; IWGCBC, 2011). Therefore, sustaining C_b sinks in coastal ecosystems will be crucial for making climate change adaptation strategies and reducing vulnerability of human coastal communities in the future (Nellemann et al., 2009; Laffoley and Grimsditch, 2009). The land-sea interaction area in coastal ecosystem is commonly termed "gray zone" in the global C cycle. The available findings and reports on C_b ecosystem are important for evaluating global earth surface C storage (Sifleet et al., 2011).

C_b biosequestration in mangroves, saltmarshes and seagrasses can capture atmospheric CO_2 and store it in plant biomass and sediments as C_b. C_b is considered as one of the most effective methods for long-term C storage (Duarte et al., 2013; Macreadie et al., 2014). C_b sink is determined by the processes and mechanisms that marine organisms absorb and use atmospheric CO_2 (Arrigo, 2004; Gonzalez et al., 2010). Marine organisms are responsible for 55% of global photosynthetic C fixation each year (Bowler et al., 2009; Bauer et al., 2013), but compared to terrestrial plants, phytoplanktonic marine organism biomass is only 0.05% of terrestrial plant biomass. The coastal ecosystem transports the fixed C_b to two adjacent ecosystems, including coastal ecosystems and the open oceans, as well as to the seabed buried in sediments in the form of humus. (Duarte and Cebrian, 1996; Duarte et al., 2005; Bouillon et al., 2008; Heck et al., 2008; Bauer et al., 2013). Once C_b is transformed into humus, it is temporarily removed from the C cycle. Donato et al. (2011) recently estimated that coastal mangroves could store up to 20 PgC, which was equivalent to roughly 2.5 times current annual greenhouse gas emissions globally. This is a striking observation, especially given the fact that mangroves only cover about 0.7% of the tropical forest area worldwide.

Recently, it has been reported that human activities have greatly modified the exchange of C and nutrients between terrestrial and coastal zones (Regnier et al., 2013). Atwood et al. (2015) further revealed that predators were helpful for biosequestration and greatly changed the coastal ecosystem C cycling based on their indirect effects on plant or microbial community composition and structure (Wilmers et al., 2012; Atwood et al., 2014). Therefore, we believe that the human activities, such as C_b fishery and aquaculture in coastal ecosystem, are able to alter food chain or increase the population of predators, and consequently enhance the C_b biosequestration in coastal ecosystems. The aims of this study are to (1) summarize the current knowledge on C_b ecosystem distribution and C_b density; (2) comprehensively evaluate global C_b sequestration potential and storage; (3) systematically estimate the contribution of C_b sink fishery to coastal C_b sequestration, and (4) provide recommendations for future C_b management strategies.

2. Methods

In this study, in order to make sure the statistical data are uniform and unbiased, the basic statistical data on the distribution and C_b sequestration of global mangroves, seagrasses and salt marshes were collected from the Food and Agriculture Organization (FAO) and International Working Group on Coastal "Blue Carbon" (IWGCBC) reports. The data on global fishery and aquaculture development were retrieved from the World Review of Fisheries and Aquaculture. The data related to C_b ecosystem and fishery and aquaculture in China were extracted from China Marine Statistical Yearbook (CMSY) and China Fishery Statistical Yearbook (CFSY). All parameters used in the study are from published journal papers, conferences papers and government reports. The global C_b storage and C_b sequestration are estimated by the following equations and the detailed description of related parameter values is given in Table 1.

The coastal C_b sequestrations are calculated using Eq. (1):

$$C_b = C_{rate} \times A_v \tag{1}$$

where C_{rate} is the C_b sequestration rate of a certain kind of vegetation; A_v is the distributing area for the corresponding C_b ecosystem.

The economic algae, shellfish and fishing C sequestration are calculated using Eq. (2):

$$C_{aquaculture} = C_a \times X_a \tag{2}$$

where X is the annual mean production; C_a is the C sequestration rate.

3. Global C_b sink estimation in coastal ecosystem

3.1. Global C_b ecosystem geographic distribution

Mangroves, seagrasses and salt marshes in coastal ecosystem are three major C_b pools, which spread across the globe. At least one of the three can be found in almost every country with a coastline (Giri et al., 2010; Pendleton et al., 2012; Siikamäki et al., 2012) (Fig. 1). Seagrass meadows often lie adjacent to mangroves and saltmarshes, which are subject to similar land-use pressures as mangroves though their much broader and different geographic range (Duarte and Chiscano, 1999; Hemminga and Duarte, 2000). Therefore, the estimation of areal coverage of saltmarshes and seagrass exist considerable uncertainty (Siikamäki et al., 2012). Barbier et al. (2011) estimated that the C_b ecosystem on mangroves, seagrasses and saltmarshes covered approximately 4.9×10^5 km^2 worldwide. Mangrove forests are coastal wetland forests that cover up to 75% of the tropical and subtropical shorelines of the world, so there are 111 countries with mangroves in the world (Siikamäki et al., 2012). Giri et al. (2010) reported that the total area of mangroves worldwide was 1.39×10^5 km^2, wherein Southeast Asia had obvious the largest mangrove area (66,687 km^2), which accounted for almost half of the total global mangroves area.

As Fig. 2 shows, we collect geographic data on the main mangroves and seagrass distribution in the top 20 countries in the world (Giri et al., 2011; Siikamäki et al., 2012). We find out that the main mangroves are concentrated on both sides of the equator and the total area for mangroves area in these 20 countries accounts for over 80% of the total area worldwide. The area of mangroves in Indonesia alone accounts for 2.7×10^4 km^2 or 19.5% of the world's total mangroves area. Followed by Indonesia, the next four countries with large mangroves area are Brazil, Australia, Mexico and Nigeria, which belong to other continents (Fig.2a). As Fig.2b shows, seagrass ecosystems are broadly distributed worldwide. The total area of seagrass is roughly estimated at 3.19×10^5 km^2. Meanwhile, there is an interesting phenomenon that mangroves mostly concentrate in developing countries around the equator, but seagrasses concentrate in both developing and developed countries (Giri et al., 2007). Southeast Asia has the largest area of seagrass. The total seagrass covered area in Southeast Asia is 8.13×10^4 km^2, which accounts for 25.4% of the world's seagrass (Fig.2b). The top four seagrass covered countries are Australia, Saudi Arabia, United States and Indonesia. The total area of seagrass in Australia is 4.11×10^4 km^2, which accounts for 12.9% of the world's total seagrass area. The salt marshes are mostly located in low temperate and high latitude area. In tropical areas, salt marshes would give way to mangroves (Allen and Pye, 1992). Chmura et al. (2003) roughly estimated that salt marshes covered 5.1×10^4 km^2 worldwide, which was in agreement with the estimation from Pendleton et al. (2012).

In China, most of mangroves distribute at Guangdong, Guangxi, Hainan and Fujian Province, and the mangroves covered areas are 323, 180, 135 and 134 km^2, respectively (CMSY, 2011). There are about 22 seagrass species distributed along China's coastal regions, which belong

Table 1
Summary of parameter and values to calculate C sequestration.

Parameters (equation)	Type	Value (World)	References	Value (China)	References
C_{rate} ((1))	Mangroves	226 (20–949) gC m^{-2} y^{-1}	Chmura et al. (2003), Duarte et al. (2005), Giri et al. (2011), Spalding et al. (2010)	444.27 gC m^{-2} y^{-1}	Duarte et al. (2005), Duan et al. (2008)
	Seagrasses	138 (45–190) gC m^{-2} y^{-1}	Charpy-Roubaud and Sournia (1990), Green and Short (2003), Duarte et al. (2005), Duarte et al. (2010), Kennedy et al. (2010)	83 gC m^{-2} y^{-1}	Duarte and Chiscano (1999), Duarte (2002), Zhang et al. (2015)
	Saltmarshes	218 (18–1717) gC m^{-2} y^{-1}	Chmura et al. (2003), Duarte et al. (2005)	235.62 gC m^{-2} y^{-1}	Duarte et al. (2005)., Duan et al. (2008)
	Corals reefs	148 gC m^{-2} y^{-1}	Duarte et al. (2005)	148 gC m^{-2} y^{-1}	Duarte et al. (2005)
A_v ((1))	Mangroves	139,170 km^2	Giri et al. (2011), FAO (2007)	827.57 km^2	CMSY (2011)
	Seagrasses	319,000 km^2	Siikamäki et al. (2012)	8276 km^2	Siikamäki et al. (2012)
	Salt marshes	51,000 km^2	Siikamäki et al. (2012), Pendleton et al. (2012)	1717 km^2	CMSY (2011), Duan et al. (2008)
	Corals reefs	112000km^2	Chen et al. (2003)	222.17 km^2	Lan and Chen (2006)
X_a ((2))	Algae	1.71×10^7 t	Yan et al. (2011a, 2011b) CFSY (2012)	1.41×10^7 t	Yan et al. (2011a, 2011b), CFSY (2012)
	Shellfish	1.26×10^7 t	CMSY (2011)	1.05×10^7 t	CMSY (2011)
	Fish catching	7.9×10^7 t	FAO (2014)	1.24×10^7 t	FAO (2014)
	Fish aquaculture	5.55×10^6 t	FAO (2014)	1.02×10^6 t	FAO (2014)
C_a ((2))	Algae	274–312 gC kg^{-1}	Yan et al. (2011a, 2011b), Zhang et al. (2005)	274–312 gC kg^{-1}	Yan et al. (2011a, 2011b), Zhang et al. (2005)
	Shellfish	62.9 gC kg^{-1}	Duarte et al. (2005)	22.7–88.7 gC kg^{-1}	Zhang (2013)
	Fish catching	3–15%	Wilson et al. (2009), Lavery et al. (2010)	11.5–12.7%	Yang et al. (2012)
	Fish aquaculture	3–15%	Wilson et al. (2009), Lavery et al. (2010)	11.5–12.7%	Yang et al. (2012)

Note: Fish aquaculture yield only include seawater aquaculture; Because it is difficult to calculate the net C sequestration rate by fish catching due to complex food chain relationship, we only deduce biological C fixation by fish from related research in the world; Because fish aquaculture mainly feed on bait, burial C from fish aquaculture can be calculated by 38–42% of fish yield reported by Song (2011), and the final C_b sink from fish aquaculture is calculated by biological C plus burial C.

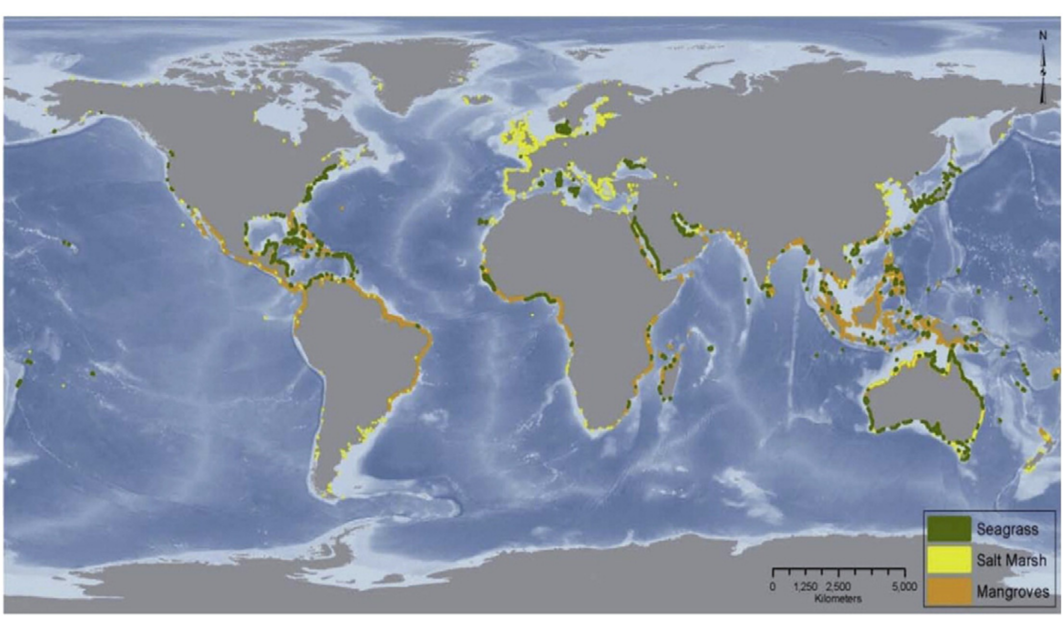

Fig. 1. Global distribution of seagrasses, saltmarshes and mangroves (Pendleton et al., 2012).

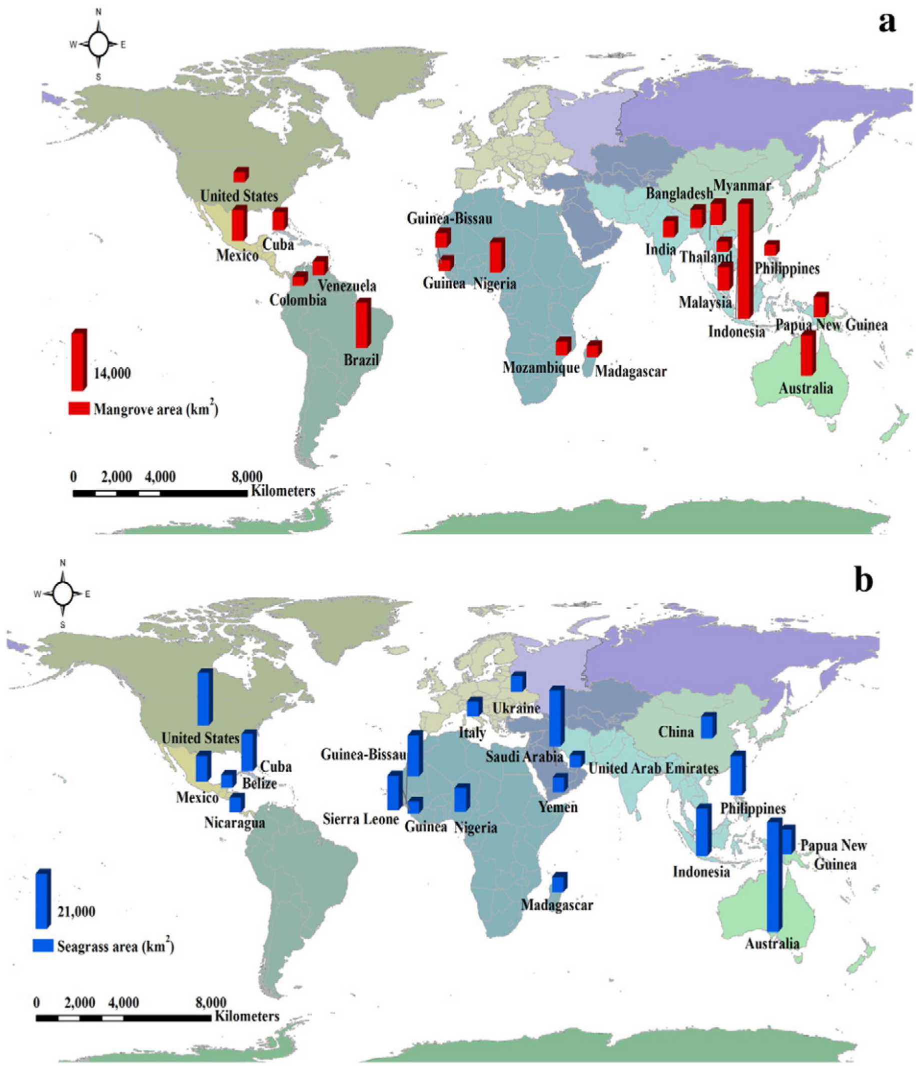

Fig. 2. Mangroves distribution (a) and seagrass distribution (b) in main top 20 countries.

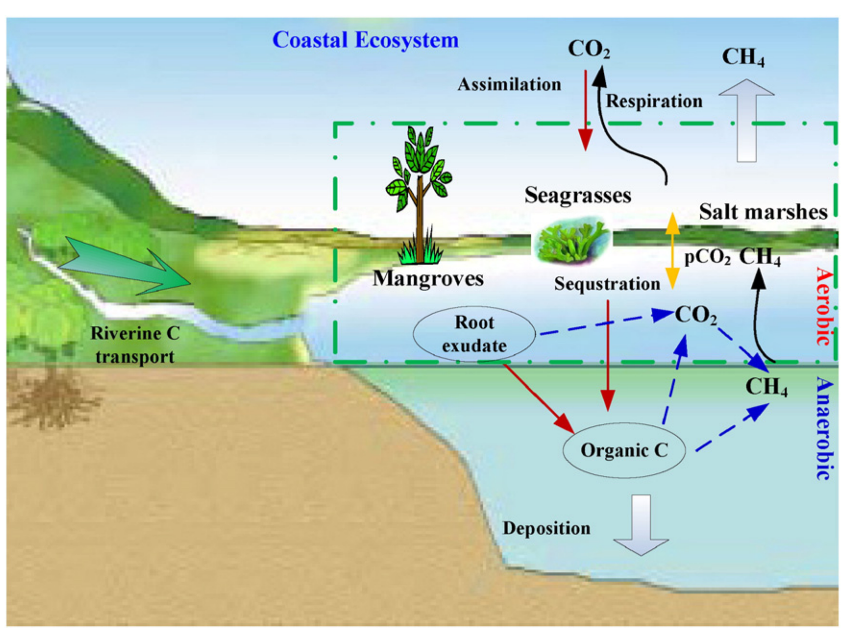

Fig. 3. C_b sequestration Process by mangroves, seagrasses and saltmarshes in coastal ecosystem.

to ten genera or four families, accounting for about 30% of known seagrass species worldwide (Zheng et al., 2013). The total distribution area of seagrass meadows in China was estimated at 8267 km^2 (Siikamäki et al., 2012), wherein Hainan, Guangdong and Guangxi province accounting for 64%, 11% and 10% of the total of area, respectively. Saltmarsh wetlands in China had an unbalanced geographical distribution.The regions that contain large amount of saltmarsh in China are the Bohai Sea, the Yellow Sea and the East China Sea. Most of the saltmarsh wetlands in China have been shrinking or exhibiting a decreasing trend. The total of area of saltmarsh wetland in China was estimated at 1717 km^2 by Duan et al. (2008).

3.2. C_b sequestration mechanism

As Fig. 3 shows, coastal ecosystem C_b is primarily sequestrated by organic matter via photosynthetic immobilization of atmospheric CO_2. Excess CO_2 would respire back into the atmosphere (Gattuso et al., 1998; Duarte et al., 2005). Mangrove roots have the capacity to simultaneously prevent saltwater intake and transport oxygen into harsh intertidal soils. Thus, their stilt and other aerial roots above the soil surface can directly uptake gases such as atmospheric CO_2 (Spalding et al., 2010). Aerial roots of mangroves can also trap suspended nutrients, peat, and C sediments from terrestrial ecosystems. This is the result of inland erosion and riverine transport (Spalding et al., 2010; Miao et al., 2011; Siikamäki et al., 2012; Miao et al., 2015; Kong et al., 2015). C_b would therefore be buried in saltmarshes within living aboveground and belowground biomass over annual to decadal timescales (Theuerkauf et al., 2015). Sources of biogenic C in salt marshes include grasses, benthic algae, and bacteria (Leonard and Luther, 1995). The riverine transport of C (as well as phytoplankton and microphytobenthos) to estuaries are potential allogenic sources of C_b into saltmarshes (Ghebrehiwet et al., 2009; Yang et al., 2015). In addition, saltmarshes can trap allogenic C from the water column (Theuerkauf et al., 2015).

Seagrass is the most productive ecosystem on Earth (Duarte and Chiscano, 1999). Its total primary production is almost equal to angiosperms and the epiphytes and macroalgae that they support (Hemminga and Duarte, 2000). Accordingly, the abundance and activity of heterotrophs in seagrass meadows would also be greatly enhanced (Hemminga and Duarte, 2000) with higher community respiration rates (Middelburg et al., 2005). Furthermore, a significant fraction of seagrass production occurs in sediments, such as roots and rhizome material (Mateo et al., 1997; Duarte et al., 2010). It should be noted that all the above C_b sink processes are strongly autotrophic in nature (Gattuso et al., 1998; Duarte et al., 2005). Some C_b sequestrated by mangroves, salt marshes, and seagrass would be buried in sediments, which can be stored over millenary timescales (Mateo et al., 1997). Based on estimations by Duarte et al. (2005), global vegetated coastal habitats accumulate approximately from 120 to 329 TgC yr^{-1} on the seafloor, thus acting as efficient C_b sinks, which accounts for at least half of the lower estimate for global C burial in marine sediments (Nellemann et al., 2009). Excess C_b would be exported into adjacent open oceans and terrestrial ecosystems (Bouillon et al., 2008).

3.3. Global C_b density in coastal ecosystem

The C_b ecosystem is different from many other ecosystems, because the majority of C_b in coastal ecosystem are immobilized and trapped in soil (Donato et al., 2011). Therefore, the estimation of C_b density in soil is an important step to understand the C_b sequestration potential in coastal ecosystem. Globally, annual coastal vegetation is around 120–329 TgC yr^{-1}, of which 13% to 25% (about 30–50 TgC yr^{-1}) is lost or buried under oceans (Duarte et al., 2005). Bouillon et al. (2008) indicated that mangroves contained substantial amounts of C_b and the C_b density in mangrove ecosystems exhibited spatial variation. Soil C accounts for the majority of C_b stored in mangrove ecosystems (Donato et al., 2011). Therefore, understanding the potential C_b density at the global level will be important to evaluate the economic potential of different locations for C_b conservation (Suratman, 2008). Due to the availability of the C_b sink geographical data for seagrass and saltmarsh, in this study we only focus on C_b density and associated geographical distribution for mangrove soil.

As Fig. 4 shows, we find out that the C density in mangrove soils is the highest in South Asia with an average density of 0.021–0.042 gC cm^{-3}, and the C density in mangrove soils is 0.025 gC cm^{-3} in East Asia. The C density in mangrove soils in the Middle East is only 0.022 gC cm^{-3}, which is the lowest as compared to other region in the world. In addition, the C densities in mangrove soils in Africa and America are 0.023–0.036 gC cm^{-3} and 0.031–0.041 gC cm^{-3}, respectively. In both Africa and America, there are significant spatial variations of the total amount of C in mangrove soil. However, the availability of spatial data on saltmarsh and seagrass at global scale are not sufficient to make a global assessment. Chmura et al. (2003) estimated that the

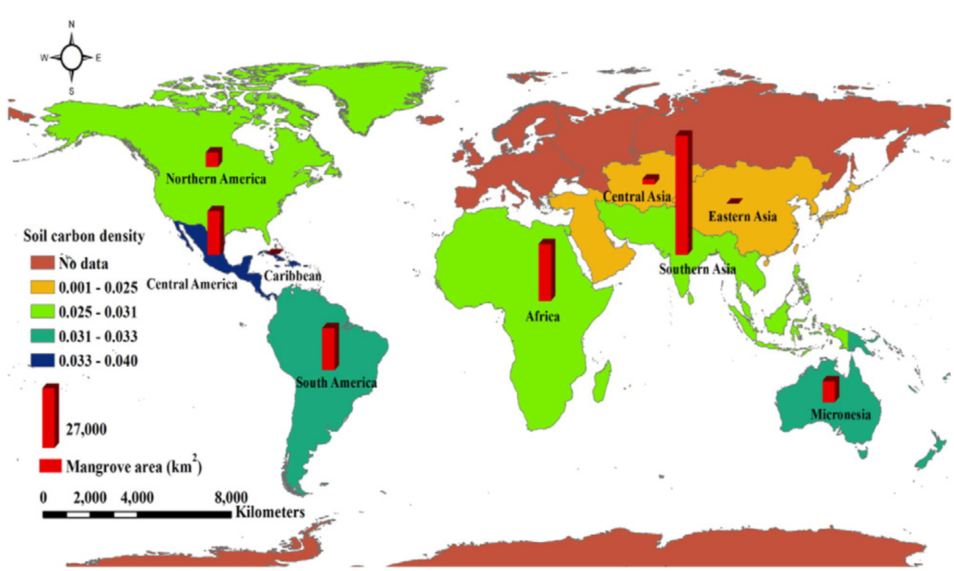

Fig. 4. Global estimation of carbon density in mangroves and mangroves area in different continent.

average C density in salt marsh was 0.039 gC cm^{-3}, which was close to that in mangroves. For C density in seagrass, Laffoley and Grimsditch (2009) reported that the soil C volume was 70 tC ha^{-1}. Duarte et al. (2005) estimated that the average C burial for seagrasses was 83 gC m^{-2} yr^{-1} and Kennedy et al. (2010) found out that there was approximately 41.5 gC m^{-2} yr^{-1} of seagrass biomass being stored in seagrass sediments.

3.4. Global estimation of C_b storage and sequestration potential

Globally, the C sequestration potential for mangroves is 31.45 TgC yr^{-1}, which accounts for 36% of the whole C_b ecosystem sequestration. In addition, seagrass beds are the primary producers (Duarte and Chiscano, 1999), with the greatest productivity sequestration potential (44.02 TgC yr^{-1}) as compared to saltmarsh and mangroves (Duarte, 2002). In microbial community, the extent of organic matter accumulation is determined by the total utilization amount of C_b that sequestered within seagrass sediments. Detrital material is the most important seagrass-derived C_b source for the microbial community in seagrass sediment, and its subsequent breakdown by microorganisms is a vital component of C cycling (Guy, 2010).

The C sequestration by C_b ecosystem in China is different from that in the globe. The global C_b sequestration potential reaches up to 86.59 Tg yr^{-1}, while China can explain 1.70% of the world's C_b sequestration. The C_b sequestration potential in China is estimated at 0.41 TgC yr^{-1} in saltmarshes and 0.37 TgC yr^{-1} in mangroves. Seagrasses are able to fix significant amounts of atmospheric CO_2 (Kaiser et al., 2005). The total seagrass covered area is about 8267 km^2 in China, which accounts for 2.6% of the world's seagrass area (Siikamäki et al., 2012). Therefore, the current study infers that the C_b sequestration potential of seagrass in China is 0.69 TgC yr^{-1}. The C_b acts as a sink and the capacity is largely dependent on the amount of organic matter accumulated within sediments following microbial transformation (Guy, 2010). In China, seagrass is the primary source for C sequestration. The total C sequestration in China equals to 47% of the whole C_b ecosystem sequestration, while the C sequestration from mangroves in China only accounts for 25% of that in the world.

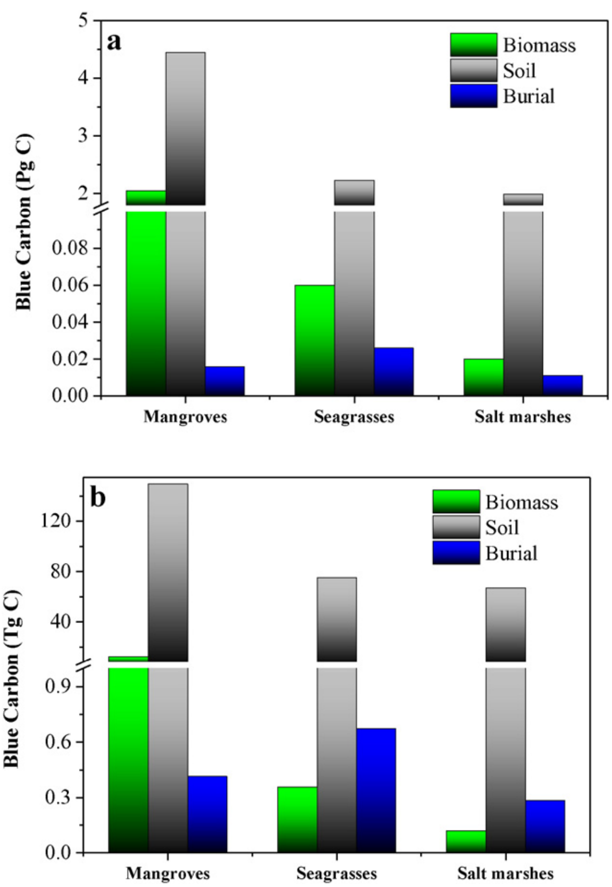

Fig. 5. Estimation of C_b stock and burial for mangroves, seagrass and saltmarshes in the world (a) and China (b).

Based on the C_b density at 1 m depth (Fig. 5), we further estimate the global C_b stored by the three C_b ecosystems. Mangroves are the primary source of C_b storage at global scale. The total stored and buried C_b in mangroves are 6.5 PgC (soil plus biomass) and 16 TgC, respectively, wherein the average burial rate is around 1.15 tC ha^{-1} (Fig.5a) (Twilley et al., 1992; Chmura et al., 2003; Kristensen et al., 2008; Donato et al., 2011) The total biomass C_b storage in saltmarshes is very small (17 TgC), and the stock density is only 1.15 tC ha^{-1}. The C_b storage density in soil for saltmarshes is 390 tC ha^{-1} and has 2 PgC of total C stock (Mitsch and Gosselink, 1993; Cebrian, 1999; Chmura et al., 2003).The C_b storage density in seagrass is approximately 72 tC ha^{-1} and the total C stock is 2.3 PgC, wherein the C_b storage in biomass and soil are 0.559 and 2.233 PgC, respectively (Duarte and Chiscano, 1999; Duarte et al., 2005; Laffoley and Grimsditch, 2009; Kennedy et al., 2010). In addition, burial C_b rate and total C stock in seagrass can reach 0.83 tC ha^{-1} and 26 TgC, respectively (Kennedy et al., 2010; Duarte et al., 2010). Therefore, the global C_b ecosystem is estimated to have a C capacity of 10.8 PgC, wherein the capacities of biomass and soil are 2.13 and 8.68 PgC, respectively. The global burial C_b in coastal ecosystem is able to reach up to 53 TgC. Based on the above statistics on the C_b ecosystem area in China, we estimate that the total amounts of C_b (biomass plus soil) pools in China are162 TgC, 67 TgC and 75 TgC in mangroves, saltmarshes and seagrass, respectively. The estimated burials C_b in mangroves, saltmarshes and seagrass ecosystem are 0.41, 0.67 and 0.29 TgC, respectively (Fig. 5b).

4. C_b sink fishery

4.1. C_b sink fishery concept

C_b ecosystem also provides a diverse array of ecosystem services such as fishery production and catching, coastline protection, pollution buffering, and high rates of C sequestration (Barbier et al., 2011). Therefore, except traditional three main C_b sink, we consider the human activities, such as algae and shellfish aquaculture, fish aquaculture and fish catching in coastal ecosystem, to be the another form of C_b sink with great potential that affects C_b ecosystem. As Fig. 6 shows, shellfish can fix inorganic C in seawater by calcification, such as HCO_3^-, and then release CO_2 to seawater and atmosphere. The amount of CO_2 directly being released to atmosphere is about 0.67 mol (Frankignoulle and Canon, 1994), and shellfish is able to absorb 2 mol of CO_2. Meanwhile, the respiration from shellfish will also release HCO_3^- to seawater. Both calcification and respiration alter inorganic C cycle in oceanic ecosystem. The C_b sink during shellfish aquaculture mainly happens during the process of predating and excreting from shellfish and catching shellfish from seawater. During predating and excreting, shellfish is feed on algae and then excrete C. The excreted C will be buried and sealed in the seafloor. The shellfish catching is considered as an C_b transfer process. The C_b sink process caused by algae aquaculture is similar with seagrass. Abundant C_b is produced through the photosynthesis by algae aquaculture, wherein part of the produced C_b is stored in the sediments and other part is removed by harvest. The CO_2 released by algae's respiration will mix with seawater and be transported back to atmosphere.

The complete C_b sink process by fish aquaculture and fishery mainly includes C transfer by catching and fish excrete processes (Fig. 6). Fish consumes microalgae, macroalgae and seagrass or predate living organism, and then excretes C to seafloor. The amount of excreted C from fish is significantly larger than that from shellfish. Coral's white calcareous skeleton also exhibits s C_b sink function by calcification and has high productivity in coastal ecosystem (Chen et al., 2003). Therefore, the

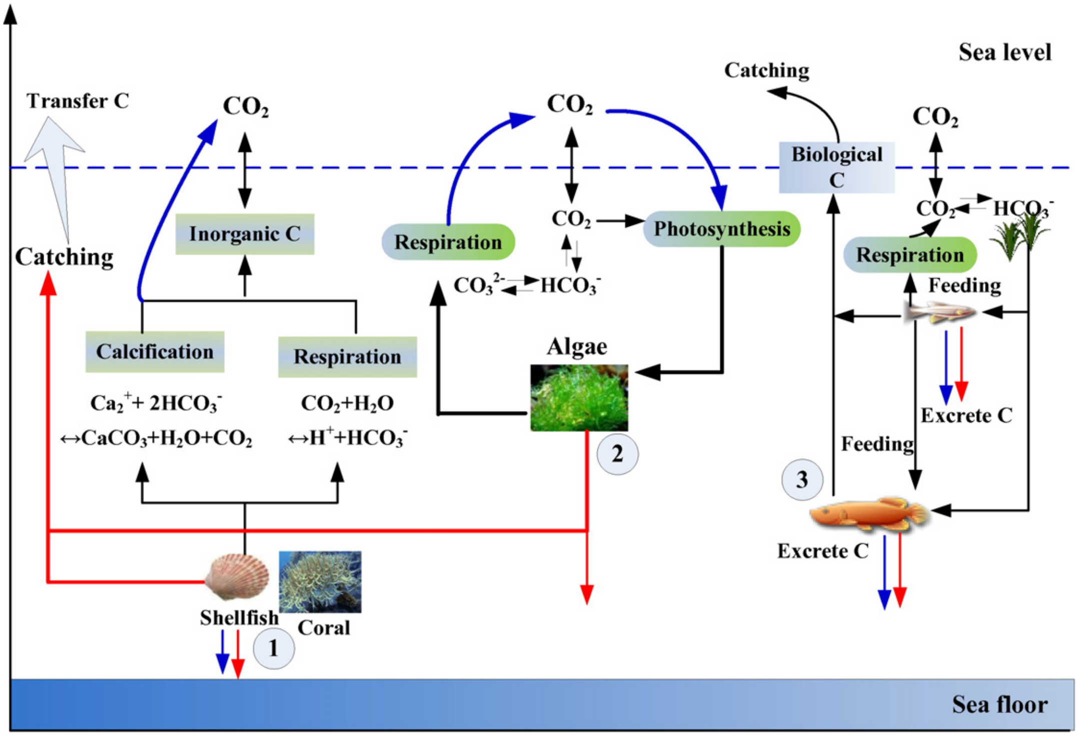

Fig. 6. Mechanism diagram of oceanic C_b sink fishery, wherein process 1 indicate the C_b sequestration process by shellfish aquaculture and coral, process 2 indicate the C_b sequestration process by algae aquaculture and process 3 indicate C_b sink process by catching or fish aquaculture. Coastal ecosystem belongs to land-sea interaction area, where C cycle is highly complicated by the significant influences of by coastal wetland ecosystem (e.g. mangroves, seagrasses and saltmarshes), human activity (e.g. fishery aquaculture), and environment change (e.g. pollutant discharge).

coral reefs' C_b sink mechanism is considered as same as that with shellfish aquaculture in this study.

4.2. Global estimation of C_b sink fishery

Duarte et al. (2009) reported that the growth rate on mariculture has been almost doubled in 10 years, and mariculture production has increased 10-fold in 30 years. Therefore, it is inferable that human activity has a considerable impact on global C_b ecosystem. The influence from human activity primarily takes place on C_b pool and C_b sequestration in coastal ecosystem. The human activity induced C_b sink fishery on C_b ecosystem is about 26.58–37.6 TgC yr^{-1}, accounting for 30.7%–43.4% of the world's traditional C_b sequestration ecosystem. Assuming that all CO_2 absorbed by economic algae comes from the atmosphere, the calculated annual C_b sequestration by economic algae worldwide is 4.65–5.30 TgC yr^{-1} (Fig. 7). The world annual average shellfish aquaculture production is 1.26×10^7 t, which might lead to a long-term confinement of approximately 0.79 TgC yr^{-1} from shells. This is because that shell matter in the upper levels of oceans is another important factor that determines the CO_2 absorb rate of seawater (Duarte et al., 2005). Shells could help enhance CO_2 uptake by dissolution in seawater, but the rate at which they dissolve is controlled by seawater acidity levels. Moreover, the process, from acidity contacts and dissolves vulnerable calcite deposits to carbonate cations arises to combine with CO_2 in surface seawater, takes extremely long time (Baliño et al., 2001).

Song (2011) reported that 1 t of fishery stocks and supplies would produce 878–952 kg of C entering seawater, accounting for 75% to 78% of the amount of total C input. According to Song (2011), 40% of the total C input would be in the dissolved form and 44% in the particulate form, which included bait residue, and excrement. According to FAO (2014), the global C_b sinks by fishery aquaculture and fish catching were 2.27–3.16 TgC yr^{-1} and 2.37–3.16 TgC yr^{-1}, respectively (Fig. 7). As a result of continues environmental pollution and climate warming, coral reefs have decreased in area from 6 to 1.12×10^5 km^2, and this trend is gradually increasing (Gattuso et al., 1998; Duarte et al., 2005). Even though coral reefs cover only 1.12×10^5 km^2 of surface area in the world, they are very productive (Chen et al., 2003). According to Duarte et al. (2005), coral reefs' C sequestration rate was about 148 gC m^{-2} yr^{-1}. Therefore, the global C_b sequestration potential of coral reefs will reach up to 16.5 TgC yr^{-1}. Due to mounting human environmental disturbances, coastal coral reef zones will become stronger atmospheric CO_2 sinks in the future.

The C_b sinks by shellfish and algae aquaculture in China are close to that worldwide. The total C_b in shellfish and algae aquaculture in China accounts for 70–80% of the world's total sequestration. China has an annual average algae production of 1.41×10^7 t (CFSY, 2012; FAO, 2014), so the calculated annual C_b sequestration by economic algae in China is estimated to be 3.84–4.36 TgC yr^{-1} (Fig. 8). Given the fact that average annual shellfish aquaculture production in China is 1.05×10^7 t (CMSY, 2011), approximately 0.24–1.1 TgC yr^{-1} of C_b sequestration will be confined within shells. The shells confined C_b accounts for 83.9% of global C sequestration by shellfish aquaculture.

The C transported by fish catching and fishing production in China accounts for 15–18% of the total amount in the world. Therefore, we estimate that the annual amount of C removed by fish catching will reach up to 1.43–1.57 TgC yr^{-1} and the total amount of annual C production by the fishery industry in China is around 0.51–0.56 TgC yr^{-1}. Currently, the coral reef area in China is only 222.17 km^2, which only accounts for 0.2% of total global coral reef C. The coral reef's area in China is mainly located in the South China Sea. In this region, approximately 55% of these coral reefs have disappeared since the 1960s (Lan and Chen, 2006). Considering the large reduction of coral reef area in China, we estimate that the C_b sequestration potential for corals reefs in China to be 0.3 TgC yr^{-1}.

Fig. 7. Global estimation of C_b sequestration potential by mangroves (a), seagrass (b) and saltmarshes(C), and human activity induced C_b sequestration including corals reefs (d), shellfish aquaculture (e), algae aquaculture (f), fish catching (g) and fishery aquaculture (h) in C_b ecosystem.

5. C_b sink management and benefit

5.1. C_b ecological service

NEA (2011) defined ecosystem services as "the outputs of ecosystems from which people derive benefits". In present, the number of studies on C_b sequestration and storage ecosystem service influenced by climate change is rapidly growing and this topic becomes a hotspot (Chung et al., 2011). C_b ecosystems in coastal ecosystem only cover 0.5% of marine areas (da Silva Copertino, 2011), but it is associated with more than 55% of marine ecosystem services. Nellemann et al. (2009) estimated that 55% of C would be removed by photosynthesis and stored by coastal ecosystems. The concept of reducing emissions from deforestation and forest degradation (REDD) are introduced to estimate the important ecosystem service on terrestrial C storage (da Silva Copertino, 2011; Luisetti et al., 2013). The use of REDD structure in estimating C_b is a viable and promising mean leading to a better understanding of C_b ecosystem services (Siikamäki et al., 2012).

Tang (2011) estimated that the amount of C sequestration by algae and shellfish in China was equivalent to the annual contribution of 5×10^5 hm^2 afforestation area in absorbing atmospheric CO_2 (Cao, 2011). Therefore, the estimated economic value of C sequestration by algae and shellfish in the last ten years was about 500 billion RMB (approximately equal to 80.65 billion USD) afforestation expense (Tang, 2011). In present study, we estimate the C_b ecosystem service by comparing with other ecosystems' C sequestration rates in China. As shown in Fig. 9, in China the C sequestration potential in forest ecosystem is far higher than that in other ecosystem. According to Fang et al., 2007, the total C sequestration potential associated with forest was 75 TgC yr^{-1}. Comparably, the C sequestration potential in coastal C_b ecosystem and C_b sink fishery are approximately 1.47 and 7.1 TgC yr^{-1}, respectively. The total C sequestration potential in C_b sink fishery is similar to that in grassland. The Chinese Karst area is considered as a new C sink, in which the C sequestration potential can reach up to 12 TgC yr^{-1}(Yan et al., 2011a, 2011b). This is mainly water in the Karst area can potentially take up a significant amount of CO_2 from the atmosphere due to the fact that the water in Karst area always contains large amount of calcium carbonate (Jiang and Yuan, 1999). Therefore, the CO_2 deposition flux usually is much smaller than that of corrosion, enabling the Karst region to act as a net carbon sink (James et al., 2006; Yan et al., 2011a, 2011b). Based on our comparison, we think the coastal C_b sink fishery and the Karst C sink will become two of the most important causes for the C sequestration potential increases in China in the future.

5.2. C_b sink estimation and management

In coastal ecosystem research, understanding impacts of human activity on ecosystems that produce C_b will aid in coastal C cycle estimation accuracy. However, there is uncertainty on how to estimate global C_b sequestration rates due to a lack of C_b areal data from ecosystems as well as differences in C burial rates between various ecosystems (Bouillon et al., 2008). Current studies have been focusing on investigating how deforestation affects global C cycles and how to effectively

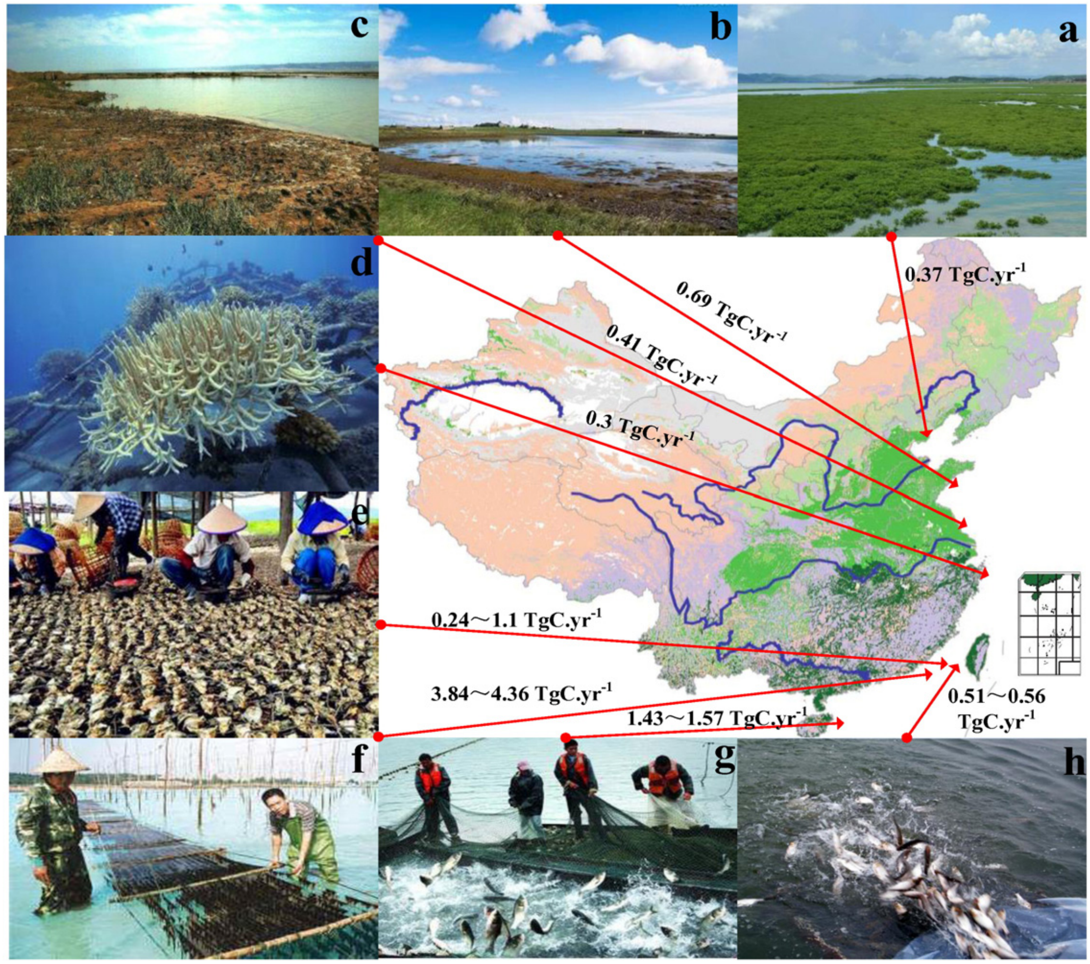

Fig. 8. Estimation of C_b sequestration potential in China. The study mainly focus on the traditional C_b ecosystem including mangroves (a), seagrasses (b) and saltmarshes (C), and human activity induced C_b sequestration including corals reefs (d), shellfish aquaculture (e), algae aquaculture (f), fishing (g) and fishery culture (h) in coastal ecosystem.

regulate CO_2 emissions. Nevertheless, it is still unclear whether C storage and sequestration in areas where land and sea interact can sufficiently absorb increases in prospective CO_2 emissions. Future research should therefore focus on strengthening C_b opportunity cost estimates as it pertains to ecosystem protection. Additionally, in some locations, such studies will also need to consider economic returns from aquaculture.

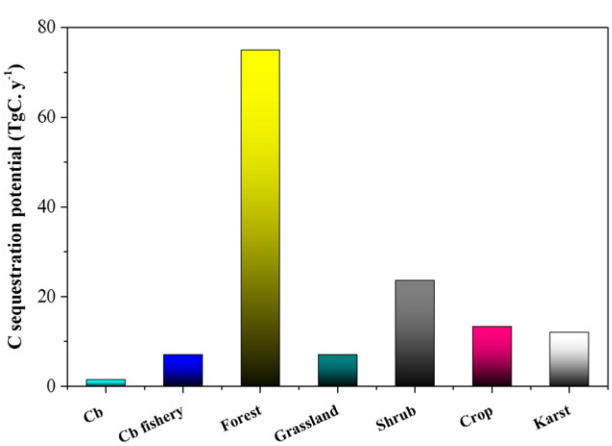

Fig. 9. C sequestration rate in different main ecosystems in China for ecosystem service evaluation, wherein C sequestration rate for different ecosystems are mainly from (Fang et al., 2007), except for Karst ecosystem from Yan et al.'s (2011a, 2011b) estimation.

Sustainable management strategies associated with C_b sink activity are critical for coastal ecosystem protection and maintenance. The objective of such management strategies would be to quantify and subsequently control C_b sequestration via algae as well as via shellfish aquaculture enterprises and coastal wetland areas themselves. As it pertains to coastal ecosystem management, it is important to differentiate between C_b sequestration, storage types, functions, and areas impacted (Everard et al., 2010). Numerous governmental bodies have already acknowledged the important role that C_b plays in ecosystems in mitigating climate change. On the other hand, it is important to accurately estimate and evaluate stored C as well as investigate ecosystem mechanisms prior to implementing plans of action to successfully protect them from degradation. The United Nations Framework Convention on Climate Change (UNFCCC) has acknowledged that there are potential benefits in maintaining stored C in ecosystems subject to C_b propagation (Siikamäki et al., 2012).

5.3. Implication for China

Considering the great economic value of coastal ecosystems, China should focus on investigating how coastal C_b propagation in ecosystem habitat conservation programs can be designed to promote the most economical and beneficial gains from fisheries and aquaculture that promote C_b sinks. Future research should also focus more attention on evaluating the economic value of C_b in ecosystems. For example, mangroves are considered to be suitable habitats for fisheries because they provide juvenile and adult fish populations with nursery habitats and food resources, respectively (Mumby et al., 2004; Mumby, 2006).

Moreover, mangroves and coral reefs are believed to be interactively correlated to fish migration and reproduction (Twilley et al., 1996). Additionally, seagrass meadow habitats are always located near coral reefs and mangroves; therefore, this type of habitat is able to provide further connectivity to those areas as well as supporting fish species that rely on reefs and mangroves (Sanchirico and Springborn, 2011).

In the future, China must develop more fisheries that promote C_b sinks based on seawater aquaculture as well as support "green" and low impact C fishery enterprises. The Chinese government must also accelerate large-scale ocean-forest engineering development programs, such as establishing seagrass meadows and deepwater macroalgae aquaculture as well as advancing novel biomass energy material. Furthermore, it is critical to establish fishery measurement standards associated with C_b sinks and, at the same time, build monitoring systems so that fisheries that promote C_b sinks (as well as their associated development potential) can be systematically evaluated and explored. Finally, such a new biological monitoring and evaluating platform will help us to develop strategies to reduce CO_2 emissions as well as potentially increase the number of C sinks.

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