J Appl Phycol (2017) 29:2363–2373
DOI 10.1007/s10811-016-1022-1

22ND INTERNATIONAL SEAWEED SYMPOSIUM, COPENHAGEN

# Carbon dioxide mitigation potential of seaweed aquaculture beds (SABs)

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Received: 11 August 2016 / Revised and accepted: 20 November 2016 / Published online: 5 December 2016


**Abstract** Seaweed aquaculture beds (SABs) that support the production of seaweed and their diverse products, cover extensive coastal areas, especially in the Asian-Pacific region, and provide many ecosystem services such as nutrient removal and $CO_2$ assimilation. The use of SABs in potential carbon dioxide ($CO_2$) mitigation efforts has been proposed with commercial seaweed production in China, India, Indonesia, Japan, Malaysia, Philippines, Republic of Korea, Thailand, and Vietnam, and is at a nascent stage in Australia and New Zealand. We attempted to consider the total annual potential of SABs to drawdown and fix anthropogenic $CO_2$. In the last decade, seaweed production has increased tremendously in the Asian-Pacific region. In 2014, the total annual production of Asian-Pacific SABs surpassed $2.61 \times 10^6$ t dw. Total carbon accumulated annually was more than $0.78 \times 10^6$ t $y^{-1}$, equivalent to over $2.87 \times 10^6$ t $CO_2$ $y^{-1}$. By increasing the area available for SABs, biomass production, carbon accumulation, and $CO_2$ drawdown can be enhanced. The conversion of biomass to biofuel can reduce the use of fossil fuels and provide additional mitigation of $CO_2$ emissions. Contributions

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of seaweeds as carbon donors to other ecosystems could be significant in global carbon sequestration. The ongoing development of SABs would not only ensure that Asian-Pacific countries will remain leaders in the global seaweed industry but may also provide an added dimension of helping to mitigate the problem of excessive $CO_2$ emissions.

**Keywords** $CO_2$ mitigation · Seaweed aquaculture bed (SAB) · Macroalgae · Blue carbon · Carbon donor · Asian Pacific region

## Introduction

Globally, carbon emissions have been increasing at an unprecedented rate leading to many negative impacts on individual species and natural ecosystems, as well as human health, infrastructure, and economies (IPCC 2014). Maintaining and improving the ability of coastal ecosystems to assimilate and store carbon is a crucial aspect of climate change mitigation. Between 1990 and 2010, estimated world-wide emissions of all major greenhouse gases reached nearly 46 × $10^9$ t, with $CO_2$ emissions in Asia among the fastest (EPA 2014). The United States, China, European Union (EU), and India are the top 4 emitting countries/regions, accounting for almost 61% of emissions (China—30%, the USA—15%, EU—10%, and India—6.5%) respectively (Olivier et al. 2015).

Vegetated coastal ecosystems, such as seagrass beds, mangroves, and tidal saltmarshes, make globally significant contributions to carbon storage in biomass and long-term sequestration in sediment deposition (Duarte et al. 2013). The carbon sequestered in both living and non-living biomass in the ocean and coastal habitats has been termed "blue carbon" (Nellemann et al. 2009; Vierros 2013; Howard et al. 2014). These ecosystems take $CO_2$ from the atmosphere via photosynthesis at the same time releasing oxygen to the air. Some carbon is released back into the atmosphere through respiration and oxidation, but a proportion of assimilated carbon remains in the form of living biomass and contributing to organic carbon stored in soils (Murray et al. 2011). The standing biomass of commercial seaweed aquaculture beds (SABs) represents additional aquatic vegetation that could enhance carbon sequestration in coastal seas. This is especially significant where SABs are located in shallow waters where the natural standing biomass of vegetation is absent or low (Mitra et al. 2014).

Seaweeds, including kelps, are important primary producers in coastal environments (Littler and Murray 1974; Smith 1981; Okuda 2008). Seaweed beds and forests, together with seagrass beds and mangrove forests, support the livelihood of millions of people and provide many ecosystem services in the coastal environments. The three-dimensional structure of natural seaweed beds/forests provides shelter to many organisms, and these beds also serve as feeding and nursery grounds for many commercially important species (Olafsson et al. 1995; Paddack and Estes 2000; Eklöf et al. 2006; Christie et al. 2009; Wattage 2011; Walsh and Watson 2011; Eklöf et al. 2012; Valderrama 2012). Natural seaweed beds/forests play very important roles in facilitating recruitment of marine organisms (Okuda 2008), absorbing excess nutrients (Fei 2004; Yang et al. 2006; Huo et al. 2012), dampening waves (Jackson 1984; Anderson et al. 1996; Lovas and Totum 2001), buffering against ocean acidification (Gao and Zheng 2010), and potentially in serving as a carbon sink for anthropogenic $CO_2$ (Hill et al. 2015).

Seaweed aquaculture is a key player in the world-wide aquaculture industry, providing 2.38 × $10^6$ t dw of global aquaculture production by volume (FAO 2014). Seaweeds are harvested for use as food, feed for aquaculture, fertilizer for agriculture, and in industrial and pharmaceutical applications (McHugh 2003). SABs cover extensive coastal areas, especially in NE and SE Asia. They provide many of the ecosystem services provided by natural seaweed stands. For example, SABs can provide a three-dimensional habitat for epiphytic organisms as well as for fishes and invertebrates (Zemke-White and Smith 2006). On top of this, given the volume of biomass produced in SABs, the potential for SABs to drawdown and fix anthropogenic $CO_2$ could also be significant (N'Yeurt et al. 2012; Chung et al. 2013). This potential role of SABs, however, has not been seriously evaluated. It should be far easier to quantify the amount of carbon sequestration by SABs than that by natural seaweed beds as the latter are spatially and temporally more variable (Fei 2004; Hill et al. 2015). Furthermore, being excellent nutrient removers, farmed seaweeds like *Pyropia* (formerly *Porphyra*) can be integrated into cultivation operations for fish that produce high nutrient loadings (Chopin et al. 1999; He et al. 2008).

Some of the carbon fixed by SABs is converted to dissolved organic carbon (DOC), which is then utilized by the bacterial community, converting DOC into dissolved inorganic carbon (DIC) through respiratory processes (Azam et al. 1983). Some of the $CO_2$ released during seaweed harvesting and processing is likely to be fixed by further uptake by newly planted sporelings/germlings. Considerable biomass can be found in wild populations of macroalgae. The contribution of wild and SAB seaweeds to carbon sinks depends on the fate of the organic material. By capturing atmospheric $CO_2$ through photosynthesis, plants, including seaweeds, can store large amounts of organic carbon in above- and below-ground biomass and can be used as bioenergy crops (Jansson et al. 2010). Seaweeds and seagrasses account for the assimilation of carbon ~1 Pg C $y^{-1}$ (Chung et al. 2011). It has been estimated that seagrasses, saltmarshes, and mangroves can capture 70% of C in the marine area (Nelleman et al. 2009, Fourqurean et al. 2012). Seaweeds utilize inorganic carbon dissolved in seawater as free $CO_2$ that diffuses in through cellular membranes from the surrounding seawater (Turan and Neori 2011) and as bicarbonate that is actively pumped into the cell via a carbon concentrating mechanism (Giordano et al. 2005; Raven et al. 2008). Moreover, the transformation by seaweeds of DIC into organic carbon by photosynthesis can decrease the $pCO_2$ in seawater (Tang et al. 2011). Through these processes the carbon sequestration in seaweed biomass can be considered as a potential mitigation measure against an increase in atmospheric $CO_2$ (Chung et al. 2011; N'Yeurt et al. 2012; Chung et al. 2013). This, however, remains a topic of considerable debate.

When macroalgae are used for animal or human foods, the $CO_2$ is simply regenerated during respiration and no net uptake of carbon occurs. Some material though can be buried in sediments or transported to the deep ocean where, even if remineralization occurs, the resulting DIC can be retained in deep oceanic waters for hundreds of years (Harrold et al. 1998; Dierssen et al. 2009; Trevathan-Tackett et al. 2015). Alternatively, if macroalgal biomass is used as a substitute for fossil fuels, this could potentially mitigate the rate of global climate change by reducing our reliance on the latter (Chung et al. 2011; N'Yeurt et al. 2012). The potential net reduction of greenhouse gas (GHG) emissions could be estimated if, for example, bioethanol from seaweeds produced in SABs is used as an alternative to gasoline from fossil fuel sources; though, GHG emitted in the biofuel production chain should be taken into account. The achievable reduction in $CO_2$ emissions may vary, depending on the species farmed and the location and growing season of the SABs. Some SABs and seagrass beds may also take up enough $CO_2$ to counter, or at least ameliorate, ocean acidification at a local scale (Rodella et al. 2015). For instance, Semesi et al. (2009) have shown that seagrass beds can maintain high pH and promote calcification in the green alga *Halimeda*.

At the 5th Asian Pacific Phycological Forum held in Bangkok, Thailand, in November 2005, a working group of the Asian Pacific Phycological Association (APPA) was established—the APPA-Asia Network—to examine the roles of SABs in $CO_2$ mitigation, especially in the Asian-Pacific region. This is in line with the Ocean Forestry Global Plan that proposed to return the concentration of atmospheric $CO_2$ to 1960's levels by 2200. With environmental, climatic, economic, political, social, and energy sustainability, "Ocean-healing Seaweed Forests" form a multi-dimensional global plan to completely reverse global warming while feeding $10 \times 10^9$ people with 200 kg of fish per year per person (N'Yeurt et al. 2012).

This review is a result of extensive discussion on this topic within the network. In 2012, seaweed aquaculture production in these Asian-Pacific countries was 97.6% of total world production. This review therefore compiles the most up-to-date data for SAB production and estimated $CO_2$ mitigation potentials in China, India, Indonesia, Japan, Malaysia, Philippines, Republic of Korea, and Vietnam, where production practices are already established, as well as in Australia and New Zealand, where these concepts are just beginning to be applied. These data are then compared with estimates from natural seaweed beds and other coastal habitats, including mangroves, seagrass beds, and salt marshes to evaluate the relative significance of SABs in the mitigation of global $CO_2$ emission.

## Materials and methods

To determine the total area used for seaweed cultivation and total annual production between 2012 and 2014, we examined data from the Food and Agricultural Organization (FAO) plus country reports from members of the APPA-Asian Network, collected in 2016. Estimates were based primarily on seaweed biomass, using formulae developed previously (Mann 1972; Zemke-White and Ohno 1999; Gevaert et al. 2008). Here, the wet weight of biomass for all seaweeds was first converted to dry weight values (10% of the wet weight was used as a conservative value) (Mann 1972; Gevaert et al. 2008; Chung et al. 2011; Roberts et al. 2015). Carbon content was assumed to be 30% of dry weight (Mann 1972; Zemke-White and Ohno 1999; Turan and Neori 2011; Arenas and Vaz-Pinto 2014; Roberts et al. 2015). The amount of $CO_2$ that could be sequestered was calculated by multiplying 'C assimilation' by the amount of $CO_2$ associated with 1 g of dry plant material (the 3.67 factor described above—Duarte et al. 2005; Pendleton et al. 2012; Mitra and Zaman 2014).

The percentage carbon content in harvested seaweed dry weight varies among and within species. For example, in *Kappaphycus*, the range of C content is from 20.7–43.1% (Widowati et al. 2012). Muraoka (2004) reported C contents in *Saccharina* to be 25–31%, *Ecklonia* 32–34%, *Sargassum* 33–37%, and *Gelidium* 36–40%. In other studies, the percentage C in *Saccharina* and *Undaria* was reported as 23.6%, 31.3% in *Gracilaria* (Fei 2004), and 27.3% in *Pyropia* (McVey et al. 2002). Accordingly, we have used 30% as an informed approximation for average C content, given the ranges cited above.

Estimating a globally relevant price on carbon sequestration is challenging (MacKay et al. 2015). Most studies calculated the benefits of carbon sequestration at between US$ 5 to 25 per tonne of $CO_{2e}$ (Fankhauser and Tol 1996; Emerton and Kekulandala 2003). The actual carbon price in the EU ETC under a price commitment is achieved by the country simply setting its carbon price to a fixed rate of US$20 (Cramton et al. 2015). The level of carbon prices in the world market was approximately US$15 to 25 in the EU ETS (Murray et al. 2011). The carbon price in the afforestation and blue carbon is estimated to be in the same range (Murray et al. 2011; Jotzo 2012; Luisetti et al. 2013; Manley 2016). The annual economic value of $CO_2$ sequestration has been estimated at approximately US$4.0 million for coral reefs and mangroves (Cesar et al. 2000), US$8.4 million for coral reefs (Samonte-Tan and Armedilla 2004), and US$2.4 million for wild seaweed beds (Vasquez et al. 2014). As it is very complex to accurately estimate the carbon price and is beyond the scope of our study, here we simply apply a conservative value of carbon as US$ 10 per tonne of $CO_{2e}$.

## Results

About 100 seaweed taxa have been cultivated in many areas around the globe but about 98% of seaweed production is accounted for by a smaller range of species from such genera as *Saccharina*, *Undaria*, *Pyropia*, *Eucheuma/Kappaphycus*, and *Gracilaria* (Turan and Neori 2011). Among Asian-Pacific countries, the major economically important seaweed groups have been used for food and feed (humans and animals), materials for industry, traditional medicine, biofertilizers, and as biofuel (bioethanol, biodiesel) (Hong et al. 2007; Phang et al. 2010).

Among the seaweeds used commercially until 2010, *Saccharina japonica* had the highest production by volume (60%) followed by other taxa such as species of *Pyropia, Kappaphycus, Undaria, Eucheuma*, and *Gracilaria*. *Pyropia* spp. are the most economically valuable seaweeds among a total global value of seaweed of US$ 6.4 × $10^9$ (FAO 2014). Recent data showed that *Kappaphycus* + *Eucheuma* have surpassed global production of *S. japonica* with about 5.5 × $10^6$ t $y^{-1}$ (FAO, 2014).

Data for SAB production were compared among APPA Network countries, incorporating information about current harvests, C assimilation, and the potential for $CO_2$ sequestration. No data were available for New Zealand and Australia because neither country was active in these efforts before 2012.

SABs yields in APPA network countries are shown in Table 1. In 2014, total annual production exceeded 2.61 × $10^6$ t dw $y^{-1}$. The highest production was in China with 1.28 × $10^6$ t dw $y^{-1}$, while the lowest was in India, at 300 t dw $y^{-1}$. Overall, estimated C assimilation was about 0.78 × $10^6$ t dw $y^{-1}$ and the potential for $CO_2$ sequestration could be 2.87 × $10^6$ t $y^{-1}$, valued at US$ 29 million based on the conservative value of carbon of US$ 10 per tonne of $CO_{2e}$. Thus carbon sequestration by SABs may provide a relatively small but significant contribution to the current world market value of seaweed aquaculture production valued at over US$ 6 × $10^9$ (Table 1).

Between 2012 and 2014, the total average production of SABs was 2.31 × $10^6$ t dw $y^{-1}$. China had the highest amount (1.11 × $10^6$ t dw $y^{-1}$) while India had the lowest (300 t dw $y^{-1}$). The total C absorption was greater than 694,636 t dw $y^{-1}$ during our survey period, and the value of potential $CO_2$ sequestration was estimated to be more than 2.54 × $10^6$ t $y^{-1}$ (Tables 2 and 3).

## Discussion

China, Indonesia, and the Philippines are the world's top three producers and account for 91.31% of global seaweed production. Production by Seaweed Aquaculture Beds (SABs) in Asian Pacific Phycological Association (APPA) Network countries increased from 20.02 Mt in 2012 to 26.13 Mt wet wt in 2014 (FAO 2016). Our review demonstrates that 694,636 t of carbon (2.54 × $10^6$ t of $CO_{2e}$) could be assimilated annually by SABs in the Asian Pacific region.

### $CO_2$ sequestration by coastal ecosystems

We compare SABs' potential $CO_2$ sequestration to terrestrial ecosystems and other blue carbon ecosystems such as

**Table 1** Estimates of algal harvests, annual carbon absorption, and potential $CO_2$ sequestration by SABs in 2014

| | Algae harvested (t ww $y^{-1}$) | Algae harvested value (1000 US$) | Algae harvested (t dw $y^{-1}$) | Estimated C assimilation (t dw $y^{-1}$) | Potential $CO_2$ sequestration (t dw $y^{-1}$) | Carbon price[a] (1000 US$) |
|---|---|---|---|---|---|---|
| China | 12,819,485 | 2,096,041 | 1,281,949 | 384,585 | 1,411,425 | 14,114 |
| India | 3000 | 98, | 300 | 90 | 330 | 3 |
| Indonesia | 8,971,463 | 1,513,253 | 897,146 | 269,124 | 987,758 | 9878 |
| Japan | 343,300 | 706,239 | 34,330 | 10,299 | 37,397 | 374 |
| Malaysia | 245,332 | 63,752 | 24,533 | 7360 | 27,011 | 270 |
| R. Korea | 1,082,027 | 485,430 | 108,203 | 32,461 | 119,131 | 1191 |
| Philippines | 1,549,576 | 256,293 | 154,958 | 46,487 | 170,608 | 1706 |
| Vietnam | 14,327 | 1863 | 1433 | 430 | 1577 | 16 |
| Total | 26,134,039 | 5,122,969 | 2,613,404 | 784,021 | 2,877,358 | 28,774 |

Source: FAO FIGIS (2016)

[a] Carbon price is US$ 10 per tonne of $CO_{2e}$

**Table 2** Estimates of algal harvests, annual C assimilation, and potential $CO_2$ sequestration by SABs, total average between 2012 and 2014

| | Algae harvested (t ww $y^{-1}$) | Algae harvested (t dw $y^{-1}$) | C assimilation (t dw $y^{-1}$) | $CO_2$ sequestration (t dw $y^{-1}$) |
|---|---|---|---|---|
| China | 11,138,780 | 1,113,878 | 334,163 | 1,226,380 |
| India | 4000 | 400 | 120 | 400 |
| Indonesia | 8,630,107 | 863,011 | 258,903 | 950,175 |
| Japan | 389,874 | 38,987 | 11,696 | 42,925 |
| Malaysia | 282,084 | 28,208 | 8463 | 31,057 |
| R. Korea | 1,076,977 | 107,698 | 32,309 | 118,575 |
| Philippines | 1,619,275 | 161,928 | 48,578 | 178,282 |
| Vietnam | 15,477 | 1548 | 464 | 1704 |
| Total | 23,156,574 | 2,315,658 | 694,636 | 2,549,498 |

Source: FAO FIGIS (2016)

mangroves, seagrasses and saltmarshes (Table 3). Temperate, boreal, and tropical forests are estimated to sequester 5096, 3599, and 4000 t $CO_2$ $km^{-2}$, respectively (Schlesinger 1997; Zehetner 2010). Annual range of soil $CO_2$ accumulation rates at saltmarshes and mangroves were 77–6287 and 73–2400 t $CO_2e$ $km^{-2}$ $y^{-1}$ (Chmura et al. 2003). Salt marshes, mangroves, and seagrasses stored 554, 510, and 304 t $CO_2e$ $km^{-2}$, respectively (Duarte et al., 2005). Globally, with total area about 509,170 $km^2$, mangroves, saltmarshes, and seagrasses altogether store about 11 × $10^9$ t C or about 42 × $10^9$ t $CO_2$ (Siikamaki et al. 2012). For example, a total of 5774 $km^2$ of blue carbon ecosystems (estimated at the Abu Dhabi workshop; seagrass, algal mat, mangrove and saltmarsh) potentially stored 3.9 × $10^6$ t C (144.8 × $10^6$ t $CO_2e$) (AGEDI 2013). When compared to wild seaweed beds, 49,939–124,849 $km^2$ of Australian temperate wild seaweed beds could store up to 109.9 Tg C (Hill et al. 2015) and 2012 $km^2$ of algal and seagrass beds along the coasts of Japan could store 2.7 × $10^6$ t C (Muraoka 2004).

It is important to include SABs in C emission schemes as they are increasing in terms of volume of production and cultivation area, whilst other important blue carbon coastal habitats (mangrove, seagrass, and saltmarsh areas) have decreased by 340,000–980,000 ha annually as a result of human pressures on coastal ecosystems (Murray et al. 2011). In the past 10-year production of seaweeds in Asian Pacific SABs has more than doubled (FAO 2016) and is projected to continuously increase. SABs can thus play an increasingly important role in C accumulation and sequestration.

## SABs in Asian Pacific countries

Despite the benefits above, SABs also have some impacts on surrounding areas such as reducing sunlight penetration, increasing siltation, and may lower seagrass biomass, shoot density, and cover area, although the impacts of SABs differ between cultivation methods.

In the Republic of Korea, SABs production from major cultivated species of *Undaria*, *Pyropia*, *Saccharina*, and *Sargassum* covered approximately 74,696 ha between 2003 and 2012, with total production of approximately 78,748 t dw $y^{-1}$. During that period, the total C assimilation is estimated to be 23,624 t $y^{-1}$, or 86,700 t $CO_2$ $y^{-1}$, and may be attributed to the regional expansion or addition of new areas of SABs, more intense cultivation, or the development of new seaweed strains for cultivation. The rise in production in Korea has become more pronounced since the 1980s because of various technical improvements, transplantation of new species of *Pyropia*, and the establishment of new grounds for cultivation (Chung 2015).

Both wild seaweed communities and SABs are important habitats that can also be considered as short-term blue carbon sinks and significant donors to long-term carbon sequestration along the coasts of all continents (Hill et al. 2015; Trevathan-Tackett et al. 2015). Countries with extensive shallow waters suitable for seaweed cultivation should be further explored for their contribution to mitigation efforts to reduce GHG emissions and existing wild seaweed beds/forests should be the focus of protection and restoration for their carbon sink mitigation potential.

Asian-Pacific countries have the capacity to increase production from SABs while improving their potential for $CO_2$ sequestration by increasing cultivation areas and developing new strains of cultivated seaweeds. China, Indonesia, Philippines, Republic of Korea, and Japan are already major suppliers of seaweed to the rest of the world. The Indonesian MMAF set a goal of preparing 60 clusters to stimulate the production of 10 × $10^6$ t wwt of seaweed by 2014 (MMAF 2014). Finally, the 10th Malaysian Plan was launched to optimize seaweed production while the 4th National Agriculture Policy (2011–2020) was enacted in order to boost the development of seaweed aquaculture programs in that country (Kaur and Ang 2009). In the period from 2010 to 2015, 900,000 ha of natural seaweed beds with a standing crop of 60–70 × $10^5$ t dw $y^{-1}$ have

**Table 3** Estimates of biomass, annual C assimilation, and potential $CO_2$ sequestration by other habitats

| Ecosystem | Area ($km^2$) | C assimilation ($t\ km^{-2}$) | $CO_2$ sequestration ($t\ km^{-2}$) | References |
|---|---|---|---|---|
| Mangrove | 139,170 | 139–7210 | 510–24,460 | Duarte et al. (2005), Siikimaki et al. (2012) |
| Saltmarsh | 22,000–400,000 | ≥218,180 | ≥800,060 | Chmura et al. (2003)* |
| Seagrass | 319,000 | 6270 | 22,988 | Siikimaki et al. (2012) |
| Forest | | | | |
| Temperate | 10,400,000 | n/a | 5096 | Schlesinger (1997), Zehetner (2010)* |
| Boreal | 13,700,000 | n/a | 3599 | |
| Tropical | 19,622,846 | n/a | 4000 | |

*Adopted from Mcleod et al. (2011)

*n/a* not applicable

been estimated as suitable for exploitation in Vietnam (Hong et al. 2007).

Although commercial seaweed production is still limited in Australia and New Zealand, cultivating seaweed in association with ocean-based finfish farming is the focus of a new research project by the South Australia government (MISA 2011). Trials are also underway for integrated multi-trophic aquaculture in Victoria, Australia. Moreover, investigation of the palatability and nutritional value of endemic Australasian seaweeds has begun. In New Zealand in 2010, the biosecurity categorization of the introduced kelp *Undaria pinnatifida,* was changed to enable cultivation and harvesting in areas where it is currently established (Barratt-Boyes 2012). Some small-scale harvesting of *U. pinnatifida* introduced to Tasmanian and Victorian waters is also occurring in Australia.

Growth in both seaweed volumes and economic value will depend upon improved efforts by the seaweed industry. The associated increase in biomass will provide economic returns to coastal communities when harvested (Baruah et al. 2006). Our estimates indicate that annually US$ 29 million could be obtained from potential $CO_2$ markets plus US$5.1 × $10^9$ from traditional seaweed markets. Although our estimate of the annual carbon price for sequestration by current SABs is modest, the potential for the ocean afforestation should not be neglected (Table 1).

## Seaweeds and SABs capabilities in $CO_2$ sequestration

Seaweeds can act as effective carbon sinks because their biomass is larger, and their turnover times are relatively longer, than those of other marine organisms such as phytoplankton. They also have higher proportions of recalcitrant carbon in their tissues that are not easily broken down (Gao and McKinley 1994; Delille et al. 2009; Trevathan-Tackett et al. 2015). Seaweeds can transform DIC via photosynthesis, thereby decreasing the $pCO_2$ in seawater. By removing a significant amount of carbon from the ocean at harvest time (Tang et al. 2011), these life forms provide potential tools for biomass production as well as $CO_2$ sequestration (Duarte et al. 2005). In addition, seaweeds acting as $CO_2$ sinks can sequester carbon within their biomass throughout their life spans (Chung et al. 2013) and beyond (Delille et al. 2009; Trevathan-Tackett et al. 2015).

Seaweeds and SABs can potentially make effective contributions to $CO_2$ mitigation because some seaweeds have cell wall structures and composition that can store carbon over the long-term by becoming a carbon donor to other ecosystems (Hill et al. 2015; Trevathan-Tackett et al. 2015) and by converting the biomass into a range of bioenergy products from biogas to liquid and solid biofuels. We discuss these roles in more detail below.

We realize that the problem with seaweeds being considered effective organisms for carbon sequestration is their short turnover time. Even though the life cycles and major accumulation of C in seaweeds are relatively limited compared to trees, a lot of $CO_2$ can be accumulated in a short time with a high productive capacity, so seaweeds are more effective as a recycling resource for fuel in which $CO_2$ accumulation and retention occur over a much longer time (Muraoka, 2004; Notoya, 2011). For example, *Sargassum* sp. turnover time in the Sargasso Sea is 10 to 100 years (Ramus 1992). A more recent study has found that not all seaweeds have short turnover times and some show potential for long-term carbon sequestration because they contain compounds that are very recalcitrant and are likely to break down slowly in sediments (Trevathan-Tackett et al. 2015).

Recently, seaweeds have been considered as contributors to coastal "blue carbon" in mitigating $CO_2$ (Chung et al. 2011, 2013; Sondak and Chung 2015; Hill et al. 2015; Trevathan-Tacket et al. 2015) because they contribute to storage of carbon by sequestration of $CO_2$ from seawater through photosynthesis and use it to increase their biomass (autochthonous carbon) that can potentially be transferred and deposited to other coastal ecosystems or the deep sea benthos (allochthonous carbon). In order for seaweeds to make significant contributions to global carbon sequestration, they must either have the capacity to directly store and accumulate carbon within their own habitat or transport biomass to receiver habitats where carbon can be effectively buried and organic material prevented from undergoing microbial mineralization (Hill et al. 2015). Seaweeds and other aquatic vegetation can be highly productive and an important source of carbon for adjacent ecosystems (Hyndes et al. 2014). In addition, seaweeds, due to their high rates of production, fragmentation, and ability to be transported, would also appear to be able to make a significant contribution as carbon donors to blue carbon habitats (Hill et al. 2015; Trevathan-Tacket et al. 2015). Carbon in the coastal ecosystems can be transported or donated to other ecosystems in the form of particulate organic carbon (POC), dissolved organic carbon (DOC), and dissolved inorganic carbon (DIC) (Hill et al. 2015), drifting seaweeds (Komatsu et al. 2008; Ito et al. 2009), and dislodgement of seaweed thalli (Hobday 2000; McKenzie and Bellgrove 2009).

Transport of DOC and POC from coastal vegetated intertidal habitats such as mangrove, seagrass, and seaweed can also occur via dissolved or particular matter, through migration of animals from intertidal to subtidal areas, and through a series of predator-prey interactions (trophic relay) (Kneib 1997; Bouillon and Connolly 2009). High rates of DOC loss through leaching occur rapidly following detachment of macrophyte leaves or thalli (Maie et al. 2006; Hyndes et al. 2012). POC and DOC along with nekton provide major vectors of carbon transfer across ecosystems within seascapes, and water movement plays a major role in facilitating transfer of carbon regardless of the vector (Hyndes et al. 2014).

Theories concerning the net transfer of carbon from intertidal to subtidal areas are dominated by concepts about carbon transfer among near-shore systems (Bouillon and Connolly 2009). Allochthonous materials such as macroalgae, terrestrial detritus, and marine-derived suspended sediment can be deposited in the intertidal systems through tidal exchanges and long-shore currents (Wolanski 1992; Bouillon et al. 2003; Adame et al. 2012). It was suggested that one of the main processes of carbon sequestration by seaweed beds is transfer of the drifting seaweeds (i.e., *Macrocystis*, *Durvillaea*, *Eisenia*, *Ecklonia*, and *Sargassum*) before sinking to benthic habitats and the offshore mesopelagic zone (Harrold et al. 1998; Ito et al. 2009; Fraser et al. 2011). Some species of seaweeds can be transported to new areas far from their origin (e.g., *Sargassum* in the Sargasso Sea) and can substantially increase their biomass in a free-floating stage, with a tendency to rapidly sink to the deep sea floor, which makes it much more efficient vehicle to carbon sequestration (Johnson and Richardson 1977; Smetacek and Zingone 2013). Thalli are eroded and dislodged whole kelp thalli form rafts on the ocean surface (Hobday 2000; Xu et al. 2016) and these are deposited as wrack along shorelines, inshore subtidal habitats, and canyons (Vetter and Dayton 1998; Orr et al. 2005; Wernberg et al. 2006; Crawley et al. 2009). Understanding how much of this macroalgal biomass gets deposited in areas conducive to long-term sequestration remains a key knowledge gap.

Animals also play an important role in carbon transfer within coastal and terrestrial ecosystems. For example, mesograzers such as gastropods are important in transferring kelp-derived carbon to higher level consumers in a range of marine ecosystems as well as coastal and terrestrial environment (Hyndes et al. 2014) and can therefore transport carbon from one ecosystem to another. Other offshore macrograzers such as dugongs, manatees, and green turtles consume large quantities of seagrass and seaweed, thus significantly transferring carbon when they migrate between shallow and deeper waters (Thayer et al. 1984). Moreover, various swimming, diving, and wading bird species prey significantly on nekton in shallow waters (Blaber 2000; Torres 2009) and cause transfer of carbon from sea to land (Hyndes et al. 2014).

Continued use of seaweeds as food will not achieve long-term $CO_2$ sequestration. Nonetheless, if some of the seaweed production can be converted to useful chemical products such as hydrocolloids/phycocolloids, alginate, agar, and carrageenan as thickening and gelling agents in food and biochemical industries, biofuels, and biochar (Turan and Neori 2011; Choi et al. 2014; Roberts et al. 2015) and thus avoid the use of fossil fuels, mitigation of $CO_2$ emissions can be achieved indirectly. Some reports suggest that macroalgae could be a useful source of such chemicals using techniques such as fast hydrothermal liquefaction (Bach et al. 2014). Other possible approaches include anaerobic digestion for methane production (Nkemka and Murto, 2010) or fermentation for bioethanol (Yanagisawa et al. 2013; Adams et al. 2015).

Plants can act as C sequestration agents and sinks in long term, in addition to their use as bioenergy crops, thereby reducing GHG emissions from fossil fuels (Jansson et al. 2010). Seaweeds can be classified as a bioenergy crop as they can produce renewable energy from biomass. The use of seaweeds as feedstock for biofuel is an emerging trend in biorefinery research as one approach to mitigation of atmospheric $CO_2$ (Bharathiraja et al. 2015). For example, *Sargassum* can be converted to biooil, biogas or biochar through pyrolysis (Kim et al. 2013b). The concept of combining bioenergy with carbon capture and storage (BECCS) has been identified as one mechanism to achieve energy production with a net negative atmospheric carbon emission (Hughes et al. 2012). The Ocean Sunrise Project in Japan has been developed with aims to combat global warming through seaweed bioethanol production by contributing an alternative energy to fossil fuel (Aizawa et al. 2007). Co-culturing macroalgae with industry flue-gas provides a holistic solution for carbon sequestration by recycling carbon and converting the biomass into a range of bioenergy products from biogas to liquid and solid biofuels (Cole et al. 2014).

Another possible solution is conversion of algal biomass into biochar that can be suitable for deep-buried storage of carbon. Long-term C sequestration can be achieved when C from above-ground biomass transfers into the soil for example as biochar or phytoliths (Jansson et al. 2010). The application of biochar to soil is proposed as a novel approach to establish a significant, long-term, sink for $CO_2$ (Farrelly et al. 2013). Seaweed aquaculture not only offers food production and hydrocolloids but also possibilities for production of biochar (Roberts et al. 2015). It was assumed by Bird et al. (2011) that biochar produced from algal feedstock may also be comparatively high in nutrients that may make algal biochar attractive for carbon sequestration as these might promote crop plant growth in soils supplemented with the char. Suh et al. (2014) found that *S. japonica* has potential for biochar and biofuel production. Biochar can be produced from a range of commercially cultivated seaweed such as *Gracilaria edulis*, *Eucheuma spinosum*, *Kappaphycus alvarezii*, *Sargassum* spp., *Undaria pinnatifida*, and *Saccharina japonica* biomass where 1.9 Mt dry wt can yield up to 0.33 Mt C $y^{-1}$ (Roberts et al. 2015).

## Conclusion

The ongoing development of seaweed aquaculture beds (SABs) ensures that Asian Pacific countries will remain leaders in the seaweed industry and in the achievement of carbon sequestration by seaweeds. If cultivation remains balanced, then the introduction of seaweed farming to additional areas will provide new standing stock to sequester carbon in those regions. Because SABs could provide important structure in coastal waters and may be considered as a key component in programs to combat climate change, their geographical coverage should be allowed to expand, enhancing the potential ecosystem services. SABs provide ecosystem services to adjacent ecosystems like reducing eutrophication effects caused by uncontrolled nutrient loading to coastal areas. Actions could be taken to reduce these impacts such as practicing sustainable and environmentally friendly aquaculture such as integrated multi-trophic aquaculture (IMTA).

We conclude that SABs can effectively contribute to $CO_2$ mitigation by becoming carbon donors to other ecosystems and converting the biomass into a range of bioenergy products from biogas to liquid and solid biofuels. This would represent a win-win strategy for coastal blue carbon ecosystems with the mitigation and adaptation measures that SABs could provide. The fate of exudation and fragments of seaweeds as a carbon sink in the deep sea should be assessed. Their strong performance to date, as described here, leads us to believe that SABs can be employed to sustain marine environments through their varied ecosystem services and provide nutrients to low-productive, adjacent coastal areas while also enhancing the economies of coastal communities.

**Acknowledgements** This work has been supported by the National Research Foundation of Korea, Marine Research Institute, Pusan National University (NRF-2013R1A1A2009359), and DIKTI Scholarship from the Indonesia Ministry of National Education and Culture for CFAS.

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