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ORIGINAL PAPER

Sedimentation and belowground carbon accumulation rates in mangrove forests that differ in diversity and land use: a tale of two mangroves

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Abstract Increased sea level is the climate change effect expected to have the greatest impact on mangrove forest survival. Mangroves have survived extreme fluctuations in sea level in the past through sedimentation and belowground carbon (C) accumulation, yet it is unclear what factors may influence these two parameters. We measured sedimentation, vertical accretion, and belowground C accumulation rates in mangrove forests from the Republic of Palau and Vietnam to examine how diversity (high-Vietnam vs. low-Palau), land use, and location (fringe vs. interior) might influence these parameters. Land use in this study was identified as disturbance and restoration for all mangrove forests sampled in Palau and Vietnam, respectively. Vertical accretion rates were significantly greater in Vietnam (2.44 ± 1.38 cm/year) than Palau mangrove forests (0.47 ± 0.08 cm/year; $p < 0.001$, $F_{1,17} = 24.96$). Vertical accretion rates were positively correlated to diversity ($R = 0.43$, $p < 0.05$). However, stronger correlations of accretion to bulk density ($R = 0.64$, $p < 0.01$) and significantly higher bulk densities in Vietnamese (0.67 ± 0.04 g/cm^3) than Palau mangroves (0.30 ± 0.03 g/cm^3; $p < 0.001$, $F_{1,17} = 54.4$) suggests that suspended sediments played a greater role in mangrove forest floor maintenance relative to sea level rise. Average vertical accretion rates were similar between naturally colonized (1.01 ± 0.10 cm/year) and outplanted sites (1.06 ± 0.05 cm/year) and between fringe (1.06 ± 0.12 cm/year) and interior mangrove (0.99 ± 0.09 cm/year) in Vietnam. In Palau, vertical accretion rates did not differ between disturbed (0.42 ± 0.11 cm/year) and undisturbed (0.51 ± 0.13 cm/year) mangrove forests and were higher in fringe (0.61 ± 0.15 cm/year) than interior sites (0.33 ± 0.09 cm/year; $p = 0.1$, $F_{1,7} = 3.45$). Belowground C accumulation rates did not differ

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between any factors examined. C accumulation rates (69–602 gC/m^2/year) were similar to those reported elsewhere in the literature and suggest that intact coastal ecosystems play an important role in the global C cycle, sequestering C at rates that are 10–20× greater than upland forests. Assuming vertical accretion rates measured using ^{210}Pb are an effective proxy for surface elevation, the Vietnamese and Palauan mangroves appear to be keeping up with current rates of sea level rise.

Keywords Mangroves · Sea level rise · Sedimentation rates · Vertical accretion · Belowground carbon accumulation · ^{210}Pb · Climate change

Introduction

Sea level rise (SLR) is the climate change impact anticipated to have the greatest impact on coastal ecosystems (Fagherazzi et al. 2012; McLeod and Salm 2006). Sea level has risen more than 5 cm over the last 20 years at an average rate of 3.2 mm/year, a rate that has nearly doubled since 1990 (Merrifield et al. 2009) and is expected to continue to increase in the future (Church et al. 2013). Higher sea levels will increase the temporal and spatial extent of flooding of coastal areas. This can alter sedimentation inputs to coastal ecosystems, tidal prisms, wave energy, period of inundation, and marsh elevation (Fagherazzi et al. 2012). In mangrove forests, these effects can result in restructuring of mangrove vegetative communities, permanent flooding, and ultimate loss of mangrove forests (Gilman et al. 2008). While mangrove forests are dynamic ecosystems that have survived extreme fluctuations in SLR over the past millennia (Alongi 2008, 2015), it is not clear how well-positioned mangrove forests are to survive current or future rates of SLR.

Recent studies suggest that most mangrove forests are keeping up with current rates of SLR (Alongi 2008; Cahoon et al. 2006; Krauss et al. 2010; Lovelock et al. 2015). This has been accomplished through: (1) vertical accretion (maintenance of forest floor elevation relative to sea level) and (2) horizontal expansion of landward and seaward edges of mangrove forests. Both are influenced by various parameters. Suspended sediments delivered to coastal wetlands via tidal and riverine inputs can have a significant and positive influence on vertical accretion rates (Krauss et al. 2010). Larger, heavier particles (i.e., sand) settle out in fringe areas of mangroves closer to the ocean or river channel, which results in fringe areas having higher sedimentation rates and sediment bulk densities (Anthony 2004; Capo et al. 2006; Walsh and Nittrouer 2004). Mangrove tree trunks, roots, and forest floor friction all slow water velocities as tidal waters flood the forest floor, reducing the water's carrying capacity and resulting in the deposition of sediments (Furukawa and Wolanski 1996; Furukawa et al. 1997). Higher sedimentation rates in fringing mangrove areas compared to areas more inland and further from the mangrove-water interface (i.e., interior) are also due to the lower elevation of fringe areas compared to interior mangrove areas. In salt marshes, lower elevations experience longer inundation periods and greater amount of time for particles to settle out of the water column (Cahoon and Reed 1995; Stoddart et al. 1989). This is likely occurring in mangrove forests as well. Storm events can then later influence the amount and redistribution of sediment within the mangrove forests (Fagherazzi et al. 2012).

Tree growth can also influence vertical accretion and horizontal expansion rates in mangrove forests. Leaf litter continually deposited to the mangrove forest floor is an important source of carbon (C) and leaf biomass is quickly incorporated into sediments through consumption by sesarmid crabs and other invertebrates (Alongi 2014; Lee 1998). Downed woody debris (i.e., branches, dead trunks), while a small portion of the aboveground C pool, can also significantly contribute to the mangrove forest floor (Alongi 2014; Kauffman and Donato 2012). Probably the most important plant contribution to vertical accretion and horizontal expansion are the high rates of belowground root growth (Cormier et al. 2015). The accumulation of subsurface refractory root material significantly contributes to vertical accretion rates and the overall long term stability of mangrove forests (Cahoon et al. 2006; Krauss et al. 2010; McKee et al. 2007).

Several reviews have suggested that an increase in mangrove tree species diversity may result in increased ecosystem services provided in those mangroves (Farnsworth 1998; Field et al. 1998; Twilley et al. 1996), although there are few explicit tests of these relationships to date. Services include increased productivity, which could potentially influence sedimentation and vertical accretion in mangrove forests.

Belowground production significantly increased within mixed plots of four mangrove species compared to plots of individual species (Lang'at et al. 2013). Different root structures can also play different roles in sediment trapping on the mangrove forest floor (Krauss et al. 2003). Settling of particles was greatest in *Rhizophora* sp., but sediment retention was greatest in *Sonneratia alba* pneumatophores. More diverse mangroves, and thus more diverse root structures, may be more resilient to SLR through increased belowground production as well as trapping and retention of sediment particles.

Human impacts can negatively impact vertical accretion and horizontal expansion such that mangrove forests may not maintain surface elevation relative to sea level or migrate inland. These impacts include altered hydrological, sediment, and nutrient loads that result from disturbances such as bridges, roads or levees that restrict hydrological connection to the ocean or rivers or from development and agriculture (Drexler and Ewel 2001; Ellison 1998). These structures can also physically hinder the ability of mangroves to expand horizontally (Semeniuk 1994). Deforestation is the human activity that probably has the greatest direct impact on surface elevation and C stocks of mangrove forests (Kauffman et al. 2014; Krauss et al. 2014). While few studies have examined the direct impacts of deforestation, these impacts are similar to those observed in forest gaps that form following hurricanes or lightning strikes. The harvesting or death of mangrove trees stops additional root growth and expansion of the root zone stops (Krauss et al. 2010; Smith III et al. 1994; Whelan 2005). Peat (i.e., fine roots, root zone) collapse occurs as exposed sediments in gaps experience increased sediment temperatures and oxidation, which collapses root tissue, accelerates decomposition rates of dead roots and organic matter (Granek and Ruttenberg 2008; Lang'at et al. 2014), and can increase leaching of dissolved C to nearby waters (Alongi et al. 2012; Mcleod et al. 2011). Sediment shear strength is also lower in gaps compared to intact mangrove forests and can increase erosion (Barr et al. 2012; Cahoon et al. 2003; Smith III et al. 1994). While seedlings can rapidly colonize and survive in gaps (Duke 2001; Ewel et al. 1998), the factors listed above that influence peat collapse can provide a positive feedback loop such that the mangrove forest floor subsides, impairing future seedling colonization (Cahoon et al. 2003).

Deforestation also results in significant reductions in C stocks (Donato et al. 2012; Kauffman et al. 2014). Harvesting obviously results in the loss of aboveground biomass and C through removal of mangrove trees. However, the loss of belowground C stocks that follows can far exceed that lost through harvesting. The construction of aquaculture ponds in Malaysia resulted in the removal of 150 tons of aboveground C ha^{-1}, but an overall loss of 750 tons of belowground C ha^{-1} (Eong 1993). Similar results have been reported from Indonesia (Sidik and Lovelock 2013). Recently, countries have invested heavily into restoring mangrove forests. Many projects involve the creation of monospecific plantations, largely *Rhizophora mucronata*, that often fail (Ellison 2000; Lewis and Gilmore 2007). An alternative approach is the ecological mangrove restoration technique that restores the hydrology of the system and allows natural recolonization by propagules from surrounding mangrove trees (Kamali and Hashim 2011; Lewis and Gilmore 2007). It is not clear if either of these restoration methods results in mangrove forests that provide similar ecosystem services as intact mangrove forests (i.e., sedimentation, C accumulation).

Fortunately, mangrove forests are resilient ecosystems that have survived extreme fluctuations in SLR over the last 65 million years and may be positioned to do so in the near future. A comparison of sedimentation rates to current rates of SLR in mangroves in the Austral-Indonesia region revealed that the majority of mangrove forests are keeping up (Alongi 2008, 2015). High island mangroves in the western Pacific are also well positioned to survive SLR, but this depends upon their location (e.g., fringe vs interior), sedimentation loads, and land use (Krauss et al. 2010). This suggests that identifying mangrove forests best positioned to survive SLR may be an effective strategy for adapting coastal areas to climate change impacts. Furthermore, understanding what biotic and abiotic factors influence accretion rates may increase our ability to more effectively manage or restore mangrove forests that are not keeping up with current rates of SLR.

Rod surface elevation tables (rSETS) and the naturally occurring radionuclide tracer ^{210}Pb are two methods commonly used to examine the ability of mangrove forests to keep up with SLR. rSETS are relatively inexpensive to install, easy to use, and can measure mm changes in surface elevation of coastal ecosystems over various time scales (i.e., months to years). However, they require repeat visits by the same person and changes in elevation can vary over time

(Krauss et al. 2010; Webb et al. 2013). Alternatively, average annual sedimentation and accretion rates can be measured over longer time periods (i.e., 100 years) using ^{210}Pb profiles from a single core collected at one time point. However, processing and analyzing sediment cores for ^{210}Pb is tedious and requires expensive alpha or gamma spectrometers to read samples (Ranjan et al. 2011; Robbins and Edgington 1975).

We used naturally occurring ^{210}Pb radionuclides to measure and compare sedimentation, vertical accretion, and belowground C accumulation rates in mangroves that differed in location (fringe vs. interior) and land use. Land use included comparing: (1) mangrove forests with roads (disturbed) to mangrove forests without roads (undisturbed) in the Republic of Palau and (2) mangroves restored by natural colonization of propagules to mangroves restored by outplanting *Rhizophora apiculata* propagules. These same parameters were compared between mangroves that differed in diversity. Palau mangroves only support 18 true mangrove (Duke et al. 1998) and thus support lower levels of tree diversity compared to southern Vietnamese mangrove forests that support 35 true mangrove species (Hong and San 1993). Our major goal was to identify mangrove forests currently keeping up with SLR. We hypothesized that sedimentation, vertical accretion, and belowground C accumulation rates would be greater in: (1) fringing mangroves closer to the mangrove-water interface than interior mangroves (Palau and Vietnam), (2) natural recolonized mangroves than monospecific outplantings (Vietnam), (3) undisturbed mangroves than disturbed mangroves (Palau), and (4) mangroves with higher levels of diversity (Vietnam vs. Palau). Measurements were made in permanent plots created as part of the Sustainable Wetland Adaptation and Mitigation Program (SWAMP) (Kauffman and Donato 2012).

Methods

Study sites

Republic of Palau

Study sites were established in six different coastal mangrove forests on Babeldaob Island in August of 2010 (Fig. 1a; Table 1). Mangrove forests ranged in

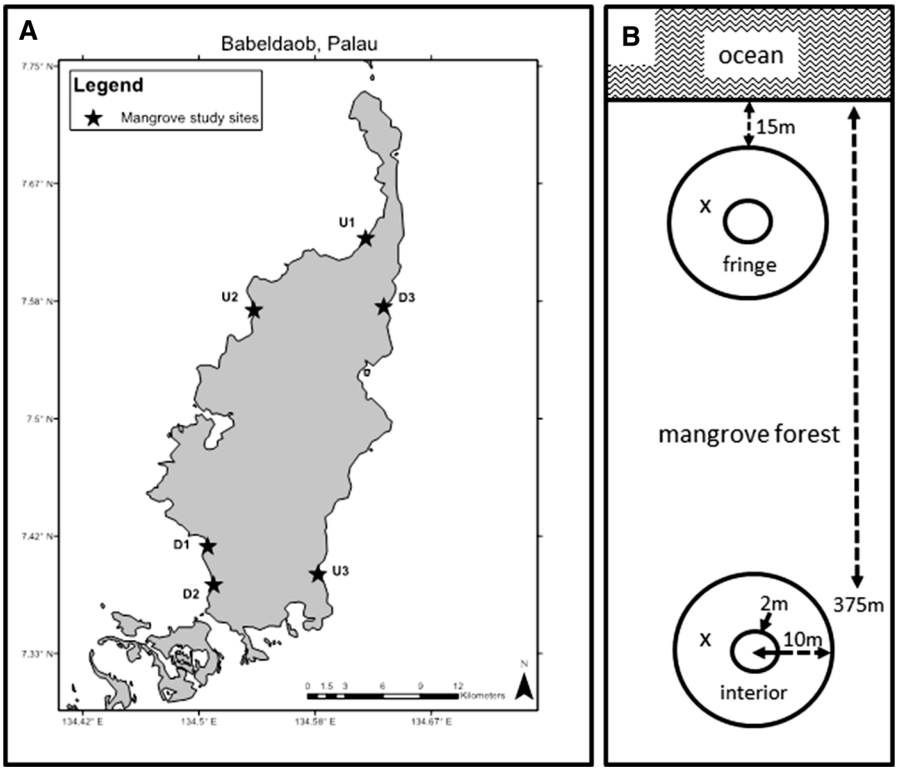

Fig. 1 **a** Map showing the locations of the study sites on the island of Babeldaob in the Republic of Palau. U sites are undisturbed and D sites are disturbed in Tables 1 and 2. **b** Layout of perpendicular transect taken from the mangrove-ocean interface. Fringe plots were located approximately 15 m from the ocean, while interior plots were located approximately 375 m. Study plots were 10-m radius with a 2-m radius nested subplot used to measure mangrove seedlings and saplings. The *x* denotes the approximate location where the sediment core was collected for ^{210}Pb analysis

Table 1 Study plots sampled from Palau and Vietnam

Plot ID	Location	Dominant species	x_projection	y_projection	Area (ha)	Species richness	Basal area (m^2/ha)	J	H′	D
Vietnam										
CG1.6	Interior	RHAP	699489	1161507	31,773	9	23.40	0.51	1.12	0.42
CG2.1	Fringe	RHAP	707723	1163670	31,773	3	22.40	0.18	0.20	0.92
KVPF1.1	Fringe	RHAP	513260	956493	11,274	4	31.70	0.25	0.35	0.83
KVPF1.6	Interior	RHAP	513178	956575	11,274	4	31.70	0.25	0.35	0.83
KVPF2.1	Fringe	RHAP	508090	955041	11,274	4	20.60	0.55	0.76	0.59
KVPF2.6	Interior	RHAP	508104	955193	11,274	4	20.60	0.55	0.76	0.59
KVPF3.1	Fringe	RHAP	511546	955573	11,274	4	28.00	0.61	0.84	0.48
KVPF3.6	Interior	RHAP	511490	955455	11,274	4	28.00	0.61	0.84	0.48
Palau										
U1B0	Fringe	RHMU	458026	843234	355	3	56.40	0.74	0.82	0.50
U1B375	Interior	RHMU	458396	843039	355	3	91.36	0.52	0.58	0.31
U2B0	Fringe	SOAL	449169	837615	147	2	34.91	0.80	0.56	0.38
U2B375	Interior	LULI	449537	837679	147	2	17.97	0.99	0.69	0.52
U3B0	Fringe	RHMU	457311	831959	307	3	77.48	0.67	0.74	0.49
U3B375	Interior	RHAP	480233	828597	307	3	47.89	0.77	0.85	0.50
D1B0	Fringe	SOAL	–	–	198	2	98.86	0.33	0.23	0.11
D1B375	Interior	BRGY	445827	818819	198	4	32.74	0.84	1.16	0.68
D3B0	Fringe	SOAL	459455	837887	160	2	63.78	0.36	0.25	0.13
D2B0	Fringe	RHMU	445978	816078	198	2	95.93	0.89	0.61	0.43
D2B375	Interior	RHAP	446348	816065	198	3	19.26	0.55	0.61	0.34

Fringe sites are located at the mangrove-water interface, while interior areas were located 150 (Vietnam) or 375 m (Palau) from the mangrove-water interface. Dominant species are mangrove trees that represented the majority of the community composition and included: RHAP (*R. apiculata*), RHMU (*R. mucronata*), SOAL (*S. alba*), and LULI (*L. littorea*). Coordinates are UTM and are in the 48 N (Vietnam) and 53 N (Palau) zones. Diversity metrics include Pielou's evenness index (J), Shannon diversity Index (H′), and Simpson's Diversity Index (D). CG represent the outplanted Can Gio plots, while KV represent the naturally restored Kien Vang plots in Vietnam. U represent undisturbed plots while D represent disturbed plots in Palau

size from 150 to 360 ha, lacked major river systems, and were adjacent to watersheds that ranged in size from 1070 to 2240 ha. The tidal range for Palau is 2 m (Golbuu et al. 2003). A fringe (~15 m from the ocean) and interior (~375 m from ocean) site were established within each of the six mangroves along a central transect running perpendicular to the mangrove-ocean interface for a total of 11 sites sampled. We were unable to sample D3B375. These transects were part of a larger mangrove forest inventory that established mangrove plots every 75 meters from the mangrove fringe to the mangrove interior. We chose 375 meters as our interior site because this distance was reached within all 6 mangrove forests selected. While all sample sites were considered relatively intact forest, three mangrove forests occurred in watersheds with a circum-island highway that has increased sediment loads to adjacent streams and mangroves (Golbuu et al. 2011). These sites will be referred to as disturbed (D). The remaining three mangrove forests were located in more remote watersheds not affected by the highway and will be referred to as undisturbed (U). Fringe sites in disturbed and undisturbed mangroves were dominated by mixed stands of *Sonneratia alba* and *Bruguierra gymnorrhiza*, while interior sites were dominated by *Rhizophora apiculata*.

Vietnam

Study sites were established in mangrove forests in the Can Gio and Cau Mau (Kien Vang Protection Forest-KVPF) provinces in the Mekong region of southern Vietnam (Fig. 2a; Table 1) in June of 2012. Mangrove

Table 2 Average values (±1SE) of bulk density, sedimentation rates (ω), vertical accretion rates (R), ^{210}Pb inventories, supported ^{210}Pb within cores (A_{supp}), percent carbon content of sediment cores, and carbon (C) accumulation rates from each core collected in Vietnam and Palau. CG represent the outplanted Can Gio plots, while KV represent the naturally restored Kien Vang plots in Vietnam

Type	Location	Plot ID	Bulk density (g/cm^3)	ω (g/cm^2/year)	R (cm/year)	^{210}Pb Inv (pCi/cm^2)	A_{supp} (pCi/g)	%C	C accumulation (gC/m^2/year)
Vietnam	Interior	CG1.6	0.64 ± 0.04	0.56	1.00 ± 0.07	7.56	0.6	4.1	227.6 ± 5.9
	Fringe	CG2.1	0.53 ± 0.02	0.58	1.11 ± 0.05	6.53	0.56	3.2	182.9 ± 72.8
	Fringe	KVPF1.1	0.70 ± 0.02	1.01	1.37 ± 0.06	6.62	0.73	3.9	388.6 ± 24.9
	Interior	KVPF1.6	0.55 ± 0.02	0.40	0.72 ± 0.06	7.81	0.69	6.8	274.5 ± 34.9
	Fringe	KVPF2.1	0.82 ± 0.02	0.80	1.01 ± 0.07	7.63	0.73	7.5	602.7 ± 30.3
	Interior	KVPF2.6	0.72 ± 0.01	0.78	1.11 ± 0.03	9.56	0.64	3.0	233.7 ± 15.4
	Fringe	KVPF3.1	0.96 ± 0.04	0.69	0.77 ± 0.05	10.57	0.63	1.7	121.0 ± 6.1
	Interior	KVPF3.6	0.73 ± 0.02	0.79	1.10 ± 0.06	11.82	0.63	2.6	206.8 ± 19.1
Palau	Fringe	U1B0	0.30 ± 0.02	0.21	0.90 ± 0.04	8.29	0.24	15.1	322.3 ± 17.0
	Interior	U1B375	0.30 ± 0.01	0.19	0.68 ± 0.03	4.23	0.18	19.9	369.7 ± 5.4
	Fringe	U2B0	0.18 ± 0.02	0.12	0.80 ± 0.05	16.35	0.25	23.6	280.6 ± 11.2
	Interior	U2B375	0.26 ± 0.01	0.04	0.22 ± 0.02	10.23	0.15	34.6	153.8 ± 3.5
	Fringe	U3B0	0.35 ± 0.02	0.07	0.22 ± 0.02	2.44	0.38	9.8	69.8 ± 6.1
	Interior	U3B375	0.28 ± 0.01	0.07	0.26 ± 0.01	4.02	0.13	21.6	157.5 ± 9.0
	Fringe	D1B0	0.24 ± 0.01	0.09	0.45 ± 0.03	8.33	0.29	15.3	141.0 ± 5.6
	Interior	D1B375	0.29 ± 0.01	0.05	0.30 ± 0.02	3.64	0.18	19.7	136.4 ± 4.8
	Fringe	D3B0	0.21 ± 0.01	0.10	0.54 ± 0.04	20.07	0.41	25.4	251.9 ± 10.1
	Fringe	D2B0	0.56 ± 0.05	0.28	0.66 ± 0.07	3.24	0.16	12.7	356.7 ± 8.7
	Interior	D2B375	0.34 ± 0.01	0.07	0.18 ± 0.01	2.76	0.11	17.7	119.5 ± 10.0

U represent undisturbed plots while D represent disturbed plots in Palau

forests ranged in size from 11,000 to 32,000 ha and were adjacent watersheds that ranged in size from 4,270,000 to 81,000,000 ha (Nguyen and Truong 2003; Ta et al. 2002). The tidal range for this region of the Mekong delta is 4 m (Hayashi et al. 2006). A fringe (~15 m from the ocean) and interior (~150 m from the ocean) plot were created within each study site along a transect running perpendicular to the mangrove-riverine/ocean interface. The interior plot distances established in Vietnam were less than those established in Palau because the transects created in Vietnam were a result of a mangrove C training workshop that only established 6 sub-plots every 25 m (Kauffman and Donato 2012) and did not allow us to sample any deeper into the mangrove forests. Furthermore, due to time constraints, we were only able to sample 2 plots from the Can Gio mangrove forest, while we were able to sample 6 plots from the Kien Vang mangrove forest.

Can Gio mangroves were heavily impacted by defoliating chemicals (i.e., Agent Orange) during the Vietnam War and then cut by adjacent villages (Hong 2001; Ross 1975). After the war, these sites were restored through a massive outplanting of *R. apiculata* (Van et al. 2014). These sites will be referred to as outplanted. The KVPF mangroves in Cau Mau were similarly impacted by the Vietnam War (Lom, pers. communication) and were also heavily logged. In contrast to Can Gio mangroves, though, KVPF mangroves were regenerated naturally without any human interference. While the majority of KVPF mangroves are in protected areas, illegal logging continues to happen but at a much smaller scale than in the past. These sites will be referred to as natural. Both outplanted and natural sites were typically dominated by *R. apiculata*.

Sediment collection and preparation

We created a 10-m and a 7-m radius sample plot within each of the fringe and interior zones in Palau and Vietnam, respectively (Figs. 1b, 2b) (Kauffman

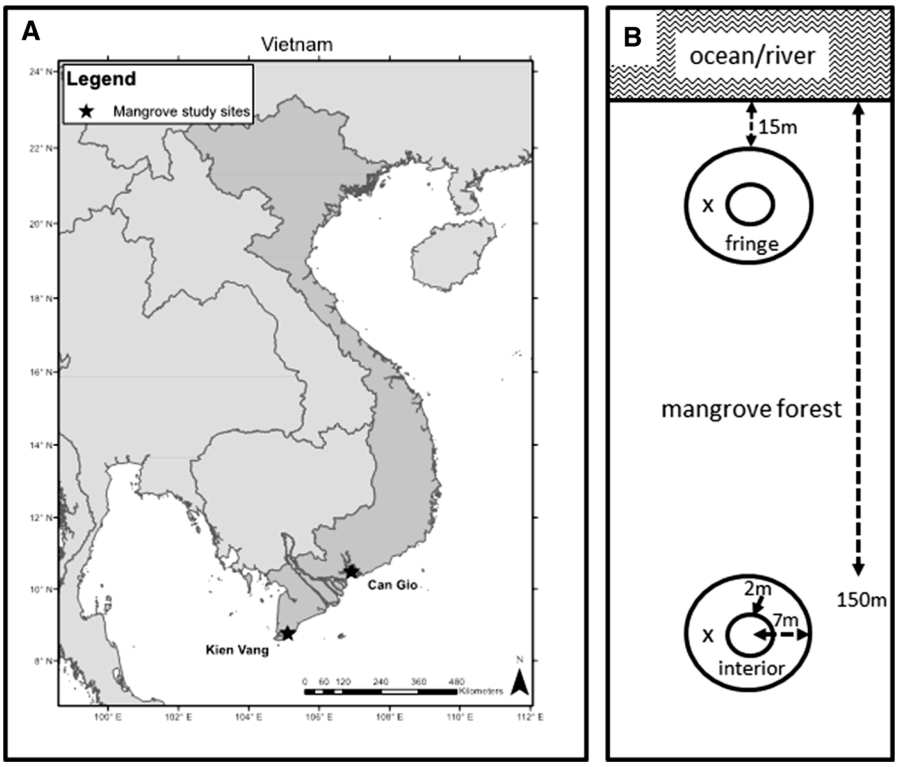

Fig. 2 **a** Map showing the locations of the study sites in the Mekong Delta of Vietnam. Can Gio (naturally restored) and Kien Vang (outplanted) are represented by CG and KV, respectively in Tables 1 and 2. **b** Layout of perpendicular transect taken from the mangrove-ocean/riverine interface. Fringe plots were located approximately 15 m from the ocean, while interior plots were located approximately 150 m. Study plots were 7-m radius with a 2-m radius nested subplot used to measure mangrove seedlings and saplings. The *x* denotes the approximate location where the sediment core was collected for ^{210}Pb analysis

and Donato 2012). Each tree species >5 cm diameter [at 1.3 m height (DBH)] was identified to species and measured. Saplings <5 cm DBH were also measured in a 2-m radius sub-plot nested within the larger 10 or 7-m radius plot. Basal density was calculated by summing up the basal area of each tree and sapling measured, converting to m^2, and dividing by the plot area (ha).

Cores were taken near the center of each plot using a 2.4-cm radius, open-faced auger (area = 18.5 cm^2) that minimized compaction of the core. The corer was carefully inserted into the sediment to minimize disturbance of the core surface, twisted, and removed. The core was then inspected for crab holes or large roots. If either were dominant, the core was thrown out and another taken. Cores were then sectioned into 2 cm intervals from 0 to 20 cm and 4 cm intervals from 20 to 60 cm. Core sections were returned to the lab where they were dried to a constant mass at 60 °C, weighed to the nearest 0.1 g, ground into a fine powder using a mortal and pestle or a Whiley^{TM} soil mill, and sieved through a 2-mm mesh sieve to remove any large pieces of wood or rocks. Bulk density was determined for each interval by dividing the total dry weight by the total sample volume.

^{210}Pb dating and calculations

The radionuclide ^{210}Po was measured from each sediment interval as a proxy for its grandfather, ^{210}Pb, assuming the two radionuclides were in secular equilibrium. ^{210}Po was extracted from sediment samples using a modified version of the digestion from Robbins and Edgington (1975). Approximately 0.5 grams of ground up and dried sediment spiked with 1 ml of radioactive recovery tracer (either ^{208}Po or ^{209}Po) was digested in a 50 ml solution of 6 N HCl acid and 30 % hydrogen peroxide for four hours. Sediments were removed by filtering through #42 Whatman paper filters and the remaining filtrate was boiled down to 5 ml. The volume of each sample was brought up to 50 ml, the pH adjusted to 0.5–1.0, and 0.1 g of ascorbic acid was added to complex out any dissolved iron present in the sample that might interfere in the plating of the Po isotopes. The Po isotopes were then plated onto polished, 2.2-cm diameter copper discs at 90 °C for 24 h (MacKenzie and Scott 1979). Copper discs were then placed in either an EG&G Ortec 576 Alpha Spectrometer or a Canberra model 7404 Quad Alpha Spectrometer

where the alpha decay of both ^{210}Po and the respective tracer were counted for 60,000 s. Replications were run every 10 samples with deviation generally less than 10 %. We chose this method over direct counts of ^{210}Pb using gamma spectrometry because alpha spectrometry of ^{210}Po: (1) requires less sediment (<0.5 g) than gamma counting ($\sim$10 g), (2) captures a better detection limit as ^{210}Pb is a weak alpha emitter (below the 46.52 keV limit of alpha detectors) and activities were generally low in our cores, and (3) requires less time to read 100s of samples.

Sedimentation rates (mg/cm^2/year) were calculated using the rapid steady-state mixing model (RSSM) that accounts and corrects for mixing (MacKenzie 2001; Robbins and Edgington 1975; Robbins et al. 1977). Activity of ^{210}Pb was first plotted against cumulative mass to correct for compaction of sediments at deeper depths and then fit with the following equation:

$$A_z = A_o e^{-\lambda(m/\omega)} + A_c \quad (1)$$

where A_z, A_o, and A_c are the depth, surface, and supported activities of ^{210}Pb, respectively, λ is the decay coefficient of ^{210}Pb (0.311), m is the cumulative mass at depth interval z (i.e., every 2 cm from 0 to 20 cm and every 4 cm from 20 to 60 cm), and ω is the sedimentation rate (g/cm^2) for that core.

The age of each interval was estimated by dividing the mass of each interval (g/cm^2) by the sedimentation rate (g/cm^2/year) of the core. The sample interval (cm) was then divided by the age of the interval to estimate vertical accretion (R) rates in cm/year, which were then averaged across intervals for each core. From this point on, vertical accretion rates will refer to cm/year and sedimentation rates will refer to mg/cm^2/year.

Carbon analysis and accumulation rates

A 0.5 subsample from each sediment interval was analyzed for total C using a Costech™ model elemental analyzer (Costech Analytical Technologies, Valencia, California). Carbon accumulation was calculated for each interval using:

$$C = \frac{R \times \%C \times mass}{SI} \quad (2)$$

where C was C accumulation (g/m^2/year), R was vertical accretion rate (cm/year), % C was percent C, mass was the mass for each interval (g/cm^2), and SI was the sample interval (cm). Carbon accumulation rates were estimated for each interval and then averaged across intervals for each core.

Statistical analysis

Pearson's product-moment correlation was used to examine which factors may be influencing sedimentation, vertical accretion, percent C and C accumulation rates in our cores (i.e., Alongi 1988; Bell et al. 1984; MacKenzie and Cormier 2012). Factors included diversity metrics [i.e., species richness, Shannon diversity Index (H′), Simpson's Diversity Index (D), and Pielou's evenness index (J)], basal area, and bulk density. Pielou's evenness is an index of how abundant each species is (where a value of 1 would represent even distribution across all species). The four diversity metrics were calculated using the DIVERSITY analysis in PRIMER 6 (Clarke 1993). We compared dependent factors among various fixed factors using separate two-way analysis of variance (ANOVA) analyses for both Palau and Vietnam. Dependent factors included: sediment bulk densities, sedimentation rates (mg/cm^2/year), vertical accretion rates (cm/year), percent C, and belowground C accumulation rates. For Palau, fixed factors included location (fringe vs. interior) and land-use (undisturbed vs. disturbed). For Vietnam, fixed factors included location (fringe vs. interior) and restoration type (natural vs. outplanted). These same dependent factors were compared across each of the individual sites in Palau and Vietnam to determine if there were significant inter-site differences within each country. All dependent factors were also compared between Palau and Vietnam using a one-way ANOVA to examine the potential influence of diversity. All analyses were run using the proc mixed command in SAS 9.1 (SAS Institute, Cary, North Carolina) with alpha values set at 0.05. The measure of dispersion is standard error.

Results

Palau

Species numbers ranged within each plot from 1-4 (2.5 ± 0.2), which resulted in an average Shannon diversity of 0.61 ± 0.09, an average Simpson's Diversity Index of 0.38 ± 0.06, and Pielou's evenness

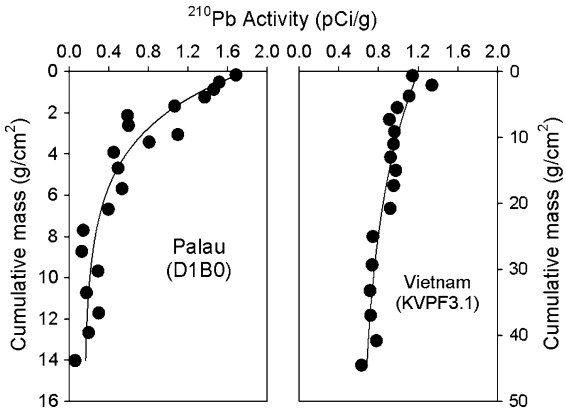

Fig. 3 Example of ^{210}Pb profiles from Palau and Vietnam. ^{210}Pb activity (pCi/g) is plotted against cumulative mass, which corrects for compaction. The slope of this line is equal to the sedimentation rate for that core (g/cm^2/year)

index of 0.63 ± 0.08. The average basal area for Palau mangrove plots was 38.0 ± 4.7 m^2/ha (Table 1).

Palau sediment bulk densities ranged from 0.18 to 0.56 g/cm^3 (Table 2). Average bulk densities did not significantly differ between disturbed (0.35 ± 0.07 g/cm^3) and undisturbed mangroves (0.28 ± 0.02 g/cm^3; $p = 0.36$, $F_{1,9} = 0.95$) or fringe (0.26 ± 0.04 g/cm^3) and interior sites (0.29 ± 0.01 g/cm^3; $p = 0.91$, $F_{1,9} = 0.12$) (Table 2). Bulk densities were similar across individual sites ($p = 0.18$, $F_{1,5} = 2.4$).

^{210}Pb supported values were relatively low, ranging from 0.11 to 0.41 pCi/cm^2 (Table 2). ^{210}Pb inventories ranged from 2.44 to 20.07 pCi/m^2. ^{210}Pb profiles were generally intact (e.g., Fig. 3) and sedimentation rates could be calculated from all 11 cores that ranged from 0.04 to 0.28 g/cm^2/year and vertical accretion rates from 0.18 to 0.90 cm/year. Average sedimentation rates did not differ between disturbed (0.12 ± 0.05 g/cm^2/year) and undisturbed sites (0.11 ± 0.03 g/cm^2/year; $p = 0.48$; $F_{1,9} = 0.56$) (Fig. 4). Average vertical accretion rates (R) were slightly lower in disturbed (0.42 ± 0.11 cm/year) than undisturbed sites (0.51 ± 0.13 cm/year; $p = 0.06$, $F_{1,9} = 5.03$) (Fig. 4). Average sedimentation and vertical accretion rates were both nearly 2× higher in fringe (0.12 ± 0.04 g/cm^2/year and 0.61 ± 0.15 cm/year) than interior sites (0.08 ± 0.03 g/cm^2/year and 0.33 ± 0.09 cm/year) (Fig. 4). This was only significant for vertical accretion rates ($p < 0.05$, $F_{1,9} = 8.58$), but not for sedimentation rates ($p = 0.19$, $F_{1,9} = 2.13$). Sedimentation rates ($p = 0.38$, $F_{1,5} = 1.3$) and vertical accretion rates ($p = 0.31$, $F_{1,5} = 1.6$) were similar among individual sites.

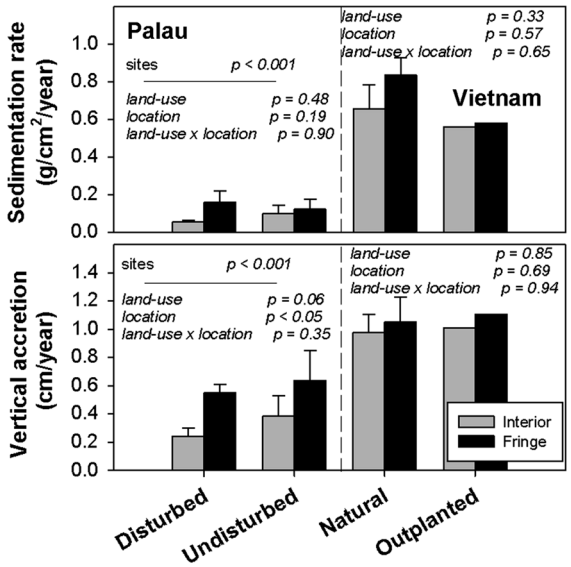

Fig. 4 Average (±1 SE) sedimentation and vertical accretion rates between fringe and interior sites sampled in Palau and Vietnam. Sites are also broken up into disturbed vs. undisturbed (Palau) and naturally restored vs. outplanted (Vietnam). Sedimentation rates are presented in g/cm^2/year (*top graph*; w), vertical accretion rates in cm/year (*bottom graph*; R). Statistical results are from the one way ANOVA between sites (Palau vs. Vietnam) and the two way ANOVAs conducted within each country where location and land use were the two independent factors tested

Percent C ranged from 9.8 to 34.6 % and did not significantly differ between disturbed (18.9 ± 2.6 %) and undisturbed sites (20.8 ± 3.4 %; $p = 0.26$; $F_{1,9} = 1.51$) or fringe (18.5 ± 3.7 %) and interior sites (22.7 ± 3.0 %; $p < 0.32$, $F_{1,9} = 1.15$) (Fig. 5). Belowground C accumulation ranged from 69.8 to 369.7 gC/m^2/year. Average belowground C accumulation rates did not significantly differ between disturbed (216.1 ± 55.3 gC/m^2/year) and undisturbed sites (225.6 ± 47.3 gC/m^2/year; $p = 0.26$, $F_{1,9} = 1.51$) fringe (231.1 ± 55.7 gC/m^2/year) and interior sites (187.4 ± 46.1 gC/m^2/year; $p = 0.31$, $F_{1,9} = 1.15$) (Fig. 5). Percent C ($p = 0.23$, $F_{1,5} = 2.01$) or C accumulation rates ($p = 0.23$, $F_{1,5} = 2.05$) did not differ among individual sites.

Vietnam

Species numbers ranged from 3 to 9 (average 4.5 ± 0.7), which resulted in an average Shannon diversity of 0.65 ± 0.11, an average Simpson's Diversity Index of 0.64 ± 0.07, and Pielou's evenness

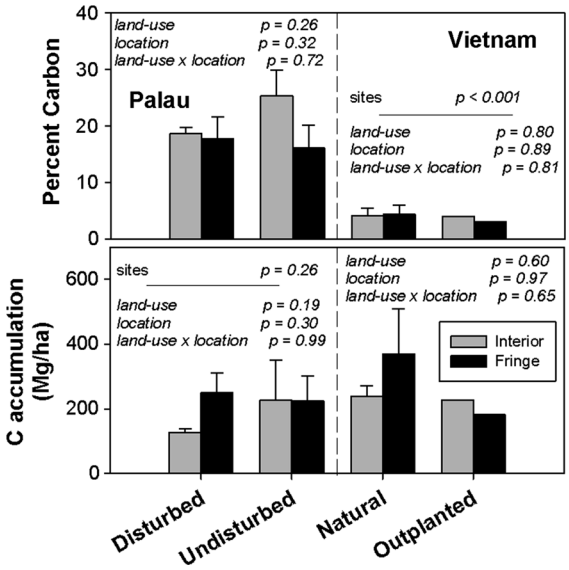

Fig. 5 Average (±1 SE) percent carbon and carbon accumulation rates between fringe and interior sites sampled in Palau and Vietnam. Sites are also broken up into disturbed vs. undisturbed (Palau) and naturally restored vs. outplanted (Vietnam). Statistical results are from the one way ANOVA between sites (Palau vs. Vietnam) and the two way ANOVAs conducted within each country where location and land use were the two independent factors tested

index of 0.44 ± 0.06. The average basal area for Palau mangrove plots was 25.8 ± 1.6 m^2/ha (Table 1).

Sediment bulk densities in Vietnam cores ranged from 0.53 to 0.96 g/cm^3 (Table 2). Average bulk densities were generally higher in naturally restored (0.75 ± 0.06 g/cm^3) than outplanted mangroves (0.59 ± 0.06 g/cm^3; $p = 0.17$, $F_{1,6} = 2.89$), although this was not significant. Average bulk densities did not significantly differ between fringe (0.75 ± 0.09 g/cm^3) and interior sites (0.66 ± 0.04 g/cm^3; $p = 0.81$, $F_{1,3} = 0.07$) or among individual sites ($p = 0.31$, $F_{1,3} = 1.8$).

^{210}Pb supported values were relatively low, ranging from 0.56 to 0.73 pCi/cm^2 (Table 2). Inventories of ^{210}Pb ranged from 6.62–11.82 pCi/m^2. Profiles of ^{210}Pb were generally intact (e.g., Fig. 3) and sedimentation rates could be calculated from eight of the nine cores collected that ranged from 0.40 to 1.0 g/cm^2/year and vertical accretion rates from 0.72 to 1.37 cm/year. Average sedimentation rates were higher in naturally restored (0.75 ± 0.08 g/cm^2/year) than outplanted sites (0.57 ± 0.01 g/cm^2/year), although not significantly ($p = 0.33$, $F_{1,6} = 1.21$) (Fig. 4). Average vertical accretion rates did not differ between naturally restored (1.01 ± 0.10 cm/year) and outplanted sites (1.06 ± 0.05 cm/year; $p = 0.86$, $F_{1,6} = 0.04$) (Fig. 4). Average sedimentation and vertical accretion rates did not differ between fringe (0.77 ± 0.09 g/cm^2/year and 1.06 ± 0.12 cm/year, respectively) and interior sites (0.63 ± 0.09 g/cm^2/year and 0.99 ± 0.09 cm/year, respectively) ($p = 0.57$, $F_{1,6} = 0.38$ and $p = 0.69$, $F_{1,6} = 0.18$, respectively) (Fig. 4). Sedimentation rates ($p = 0.38$, $F_{1,3} = 1.3$) or vertical accretion rates ($p = 0.95$, $F_{1,3} = 0.06$) did not differ among individual sites.

Percent C ranged across Vietnam sites from 1.7 to 7.5 %. Average percent C values did not differ between naturally restored sites (4.3 ± 1.0 %) and outplanted sites (3.6 ± 0.5 %; $p = 0.0.80$, $F_{1,6} = 0.08$) or fringe (4.1 ± 1.2 %) and interior sites (4.1 ± 0.9 %; $p = 0.89$, $F_{1,6} = 0.02$) (Fig. 5). Belowground C accumulation ranged across Vietnam sites from 120.9 to 602.7 gC/m^2/year. Average belowground C accumulation rates were generally higher in naturally restored (304.5 ± 69.6 gC/m^2/year) than

Table 3 Correlation coefficients from the Pearson's correlation analyses comparing potential relationships across sedimentation rates (ω), vertical accretion rates (R), percent carbon in sediments, and carbon accumulation rates and various diversity, sediment, and plot characteristics

Parameter	ω (g/cm^2/year)	R (cm/year)	Carbon (%)	Carbon accumulation (gC/m^2/year)
Species richness	0.39	0.25	**−0.52***	0.12
Shannon Diversity (H′)	0.01	−0.15	−0.08	−0.08
Simpson's Diversity (D)	**0.51***	**0.43***	**−0.48***	0.11
Pielou's evenness (J)	−0.32	−0.37	0.36	−0.14
Basal area (m^2/ha)	−0.36	−0.21	0.22	0.00
Bulk density (g/cm^3)	**0.89*****	**0.64****	**−0.83*****	0.31

Bold values represent significant correlations (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$)

outplanted sites (205.2 ± 22.3 gC/m^2/year) as well as in fringe (323.8 ± 109.2 gC/m^2/year) and interior sites (235.6 ± 14.2 gC/m^2/year), although not significantly ($p = 0.60$, $F_{1,6} = 0.32$ and $p = 0.97$, $F_{1,6} = 0.01$, respectively) (Fig. 5). Percent C ($p = 0.38$, $F_{1,3} = 1.35$) or C accumulation rates ($p = 0.28$, $F_{1,3} = 1.98$) did not differ among individual sites.

Between site comparisons

Average sediment bulk densities were significantly and 2× greater in sediment cores from Vietnam (0.67 ± 0.04 g/cm^3) than Palau (0.30 ± 0.03 g/cm^3; $p < 0.001$, $F_{1,17} = 54.4$). Average sedimentation and vertical accretion rates were also both significantly and nearly 5× higher in Vietnam (0.71 ± 0.07 g/cm^2/year and 2.44 ± 1.38 cm/year) than Palau mangroves (0.11 ± 0.02 g/cm^2/year and 0.47 ± 0.08 cm/year) ($p < 0.001$, $F_{1,17} = 88.29$ and $p < 0.001$, $F_{1,17} = 24.96$, respectively) (Fig. 4). Percent C content was nearly 5× greater in Palau (19.6 ± 2.1 %) than Vietnam sediments (4.4 ± 0.7 %; $p < 0.001$, $F_{1,17} = 38.29$) (Fig. 5). Belowground C accumulation rates were greater in Vietnam mangroves (302.4 ± 52.5 gC/m^2/year) than Palau mangroves (214.5 ± 31.6 gC/m^2/year), although not significantly ($p = 0.26$, $F_{1,17} = 1.34$) (Fig. 5).

Sedimentation and vertical accretion rates were positively and significantly correlated to Simpson's Diversity Index ($R = 0.51$, $p < 0.05$ and $R = 0.43$, $p < 0.05$, respectively) and to bulk density ($R = 0.89$, $p < 0.001$ and $R = 0.64$, $p < 0.01$, respectively) (Table 3). Percent C content of sediment cores was negatively and significantly correlated to species richness ($R = -0.52$, $p < 0.05$), Simpson's Diversity Index ($R = -0.48$, $p < 0.05$), and bulk density ($R = -0.83$, $p < 0.001$). There were no correlations between belowground C accumulation rates and any variables examined.

Sedimentation rates versus sea level rise

Altimetry has measured a SLR rate from 1993 to 2010 for Palau of 9 mm/year (CSIRO 2010), while a tide gage in Malakal, Palau only reported a 1.5 mm/year rate from 1969 to 2000 (Church et al. 2006). Tide gages along the coast of Vietnam measured a SLR rate from 1993 to 2008 of 3.1 mm/year (Nguyen 2009). When vertical accretion rates are converted to mm/year, all mangroves sampled in Cau Mau and Can Gio appear to be keeping up with current rates of SLR (Fig. 6). Under the 9 mm SLR estimate for Palau, only 1 of the 11 mangroves appears to be keeping up with current SLR. Under the 1.5 mm SLR scenario, all 11 mangroves appear to be keeping up with SLR (Fig. 6).

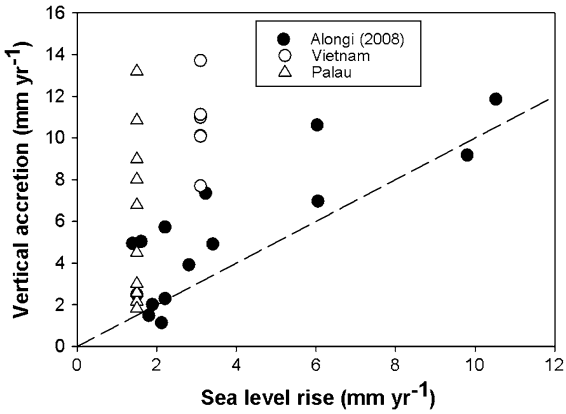

Fig. 6 Vertical accretion rates (mm/year) measured in this study and in Alongi (2008) versus current rates of sea level rise. Sea level rise values for Vietnam were obtained from Nguyen (2009). SLR presented from Palau are extreme values reported by CSIRO (2010) and were obtained using satellite altimetry

Discussion

Sea level rise is the climate change factor expected to have the greatest influence on the future distribution of mangrove forests and the greatest negative impact on mangrove trees. Fortunately, mangrove forests are dynamic ecosystems that have kept up with past sea level changes (Alongi 2008, 2015), albeit at rates that may not be as great as those in the next century. Mangrove forest survival is affected by their ability to expand vertically and horizontally, processes that are significantly influenced by sedimentation and belowground root growth. We saw no differences when we compared sedimentation or vertical accretion rates as well as belowground C accumulation rates between disturbed versus undisturbed mangroves in Palau or naturally restored versus outplanted mangroves in Vietnam or among mangrove plots within either country. Sedimentation and vertical accretion rates as well as belowground C accumulation rates were generally higher in fringe sites nearer the mangrove-open water interface, but this was only significant for vertical accretion rates measured in Palau. The lack of

significant differences is likely due to the high within site variability in sedimentation and vertical accretion rates, which has been reported in other mangroves (Lynch et al. 1989; Ranjan et al. 2011). The lack of differences between outplanted and naturally restored plots in Vietnam may also have been due to our unbalanced design and low replication. Finally, sedimentation and vertical accretion rates were significantly greater in more diverse Vietnam mangrove forests than less diverse Palau mangrove forests, although this may have been more influenced by differences in suspended sediments in waters adjacent to mangroves in Vietnam vs. Palau. Related to this argument is that these sites differ geomorphologically. Vietnam mangrove forests were deltaic coastal systems that were inundated by large rivers with high suspended sediment loads. Palau mangroves were oceanic coastal ecosystems inundated by marine waters with lower suspended sediment loads. Carbon content in sediments was greater in Palau than Vietnam, but there were no differences in belowground C accumulation rates between these two countries. This was due to the higher sedimentation rates in Vietnam.

Potential drivers influencing sedimentation patterns

Higher sedimentation rates in fringe vs interior areas of mangroves were likely due to plant structures reducing water velocity as mangroves are inundated by flooding tides and stagnant zones are created between roots and trunks. While we did not measure this, decreased velocity of water flowing through plant structures decreases carrying capacity of sediments and sediments are deposited near the wetland-water interface (Furukawa and Wolanski 1996; Furukawa et al. 1997). The greater differences observed between fringe and interior in Palau was likely due to the greater distance between these sites in Palau (375 m) than in Vietnam (150 m).

Significantly higher bulk density and lower C content of cores collected from Vietnam suggest that sedimentation and vertical accretion rates in the Kien Vang and Can Gio mangroves are influenced by sediment loads delivered by adjacent riverine ecosystems. This was also evident by the strong correlation between sedimentation and vertical accretion rates and bulk density values. Sediment loads are also important in the maintenance of the elevation of high island deltaic mangroves (Krauss et al. 2010), but the higher C content of cores from Palau suggests that belowground root growth may play a larger role in elevation of oceanic mangrove forests along island coasts. This has also been suggested in mangrove forests on Caribbean islands (McKee et al. 2007). Furthermore, coastal mangroves in Palau lacked major river systems flowing through them and were adjacent to smaller watersheds (Table 1) compared to Vietnam. Thus, they received smaller sediment loads (0.9–7.5 g/m^3) (e.g., Golbuu et al. 2011) compared to the waters adjacent to Mekong Delta mangroves of Vietnam (16–27 g/cm^3) (Lovelock et al. 2015).

It is unclear what factors have the greatest influence on sedimentation, vertical accretion, and C accumulation rates in mangrove forests. Species assemblage plays an important role, where sedimentation can be greater in mangroves with complex trunk and root structures (e.g., *Rhizophora* sp. prop roots) compared to simpler forms, such as single trunked species (e.g., *Ceriops* sp.) (Furukawa and Wolanski 1996). Sediment deposition has also been enhanced in areas with *Rhizophora* sp. prop roots compared to areas with *Sonneratia alba* pneumatophores or bare control areas (Krauss et al. 2003). However, when subsurface changes were examined using sediment pins, they found the opposite pattern. Furthermore, there may be a threshold where certain root densities may facilitate turbulence-induced erosional processes (e.g., Furukawa et al. 1997).

Correlation of sedimentation and vertical accretion rates, percent C, and C accumulation across mangroves that varied in diversity supports the idea that species diversity may potentially increase sedimentation and vertical accretion rates. Sedimentation rates were positively correlated to Simpson's diversity index. The lack of correlation to the Shannon Index, which is more sensitive to changes in rare species, was likely because study plots were only dominated by a few species. While these results are interesting, caution should be used in their interpretation as only a small data set was compared from two areas that only slightly differed in diversity. Vietnam plots only had 2× more species than Palau plots. Alternatively, the relationship between sedimentation and diversity could be compounded by differences in the physical conditions between Vietnam mangrove forests (e.g., higher sediment loads in deltaic mangrove forests)

than the Palau mangrove forests (e.g., lower sediment loads in oceanic coastal mangrove forests) that could result in higher sediment loads and sedimentation rates. Higher sedimentation can also provide ideal habitat to support a greater number of tree species (Friess et al. 2011). Further analyses on larger data sets currently being collected across a greater diversity gradient in the region will provide additional insight into these results.

It is not clear why diversity measures were negatively correlated to sediment C content. It could simply be an artifact due to dilution of C content by inorganic sediments in Vietnam cores, higher root growth occurring in Palau mangroves, or deposition of organic marine material in coastal Palau mangroves (Bouillon et al. 2003). Alternatively, belowground root growth and thus inputs of C to these systems may be higher in less diverse mangroves as there may be less competition. Increased diversity of an upland forest increased belowground interspecific competition and thus root growth or shoot:root ratios of some species, but not others (Leuschner et al. 2001). However, belowground interspecific interactions typically result in increased root growth (Bolte and Villanueva 2006; Lang'at et al. 2013; Schmid 2002). It's not clear if belowground interspecific competition is also occurring in mangroves, although interspecific difference in root production has been reported (Gleason and Ewel 2002). Finally, there may have been differences in nutrient inputs in these two mangroves that we did not measure. Increased nutrients can increase belowground root growth and thus belowground C content (McKee 1995; McKee et al. 2007).

Belowground C accumulation in mangroves

Belowground C accumulation values were quite different across the 19 Vietnam and Palau sediment cores. When values were averaged across treatments, there were no major differences, despite differences observed in sedimentation rates and in C content of sediments. The lower C content in Vietnam cores was compensated by the much higher sedimentation rates, which resulted in similar C accumulation rates between Palau and Vietnam.

The range of values reported from our sites (Table 2) were similar to average values reported from mangroves by Alongi (2014) (174 ± 23 gC/m^2/year), Lovelock et al. (2014) (72 ± 12 gC/m^2/year), and McLeod et al. (2011) (226 ± 38 gC/m^2/year). Our values were also similar to average values reported from salt marsh ecosystems by Lovelock et al. (2014) (112 ± 58 gC/m^2/year) and McLeod et al. (2011) (218 ± 24 gC/m^2/year). Belowground burial for seagrass meadows is also high (Fourqurean et al. 2012), but generally lower (138 ± 38 gC/m^2/year) than values we observed in our systems. These results highlight the importance of healthy coastal ecosystems and the important role they play relative to terrestrial ecosystems at removing carbon dioxide (CO_2) from the atmosphere (Mcleod et al. 2011), storing it for long periods of time (McKee et al. 2007), and potentially offsetting increased CO_2 levels in the atmosphere.

Mangroves versus SLR

The vertical accretion rates we report here for Vietnam and Palau were similar to sedimentation rates measured in Australia using ^{210}Pb (Alongi 2008) (Fig. 6), but higher than those reported from Mexico and Florida (0.1–0.4 mm/year) (Lynch et al. 1989). Our rates are also similar to surface elevation rates measured using surface elevation tables (SETs) in Micronesia (2.9–20.9 mm/year) (Krauss et al. 2010), the Caribbean (0.7–3.5 mm/year) (McKee et al. 2007), and the United States (1.1–9.9 mm/year) (Cahoon 2014). Vertical accretion rates calculated from ^{210}Pb profiles should not be confused with surface elevation rates determined from rSETS or SETS as ^{210}Pb profiles typically ignore subsurface processes (e.g., peat collapse in depths >1 m) that can influence surface elevation of mangroves (Cahoon et al. 2003). However, the fact that our rates are within the range reported from other mangroves using SETS suggests ^{210}Pb profiles may be an accurate proxy for accretion rates and can immediately yield average accretion rates that have occurred over the last 100 years [as opposed to making repeated measures over time that vary using rSETS (Webb et al. 2013)]. Future research will examine the relationship between rSETS and ^{210}Pb profiles at our Palau sites, where rSETS have also been installed and are only beginning to measure surface elevation.

If we assume that vertical accretion measures are good proxies for surface elevation, comparisons of our

results to current rates of SLR suggest that most of the mangroves we have sampled are keeping up with the current rates of SLR. In Vietnam, current sedimentation rates are likely influenced by higher suspended solids in the water compared to Palau (Hoa et al. 2007). This is of particular concern as nearly 200 dams have been proposed to be built on the Mekong River, which will likely impact sediment loading to coastal ecosystems, including mangrove forests (Walling 2008). While several dams have been built in the region, sediment loads have not significantly decreased in the middle and lower Mekong region, although any dam impacts may have been offset by increased land use (Fu et al. 2008; Lu and Siew 2006; Walling 2008). Furthermore, the available data have significant limitations in terms of continuity and length of records (Walling 2008). One modelling study reported that the construction of additional reservoirs may increase trapping by more than 50 % of the modeled total basin sediment load (Kummu et al. 2010). This suggests that the increased number of dams in the upper reaches of the Mekong delta threaten both the natural delivery of sediment (Hoa et al. 2007; Kummu and Varis 2007) and the potential for Mekong mangrove forests to keep up with future rates of SLR. Additional monitoring data is needed to determine if and how an increase in the number of dams will impact sediment dynamics.

For Palau, if sea level rises at 9 mm/year (CSIRO 2010), only one mangrove forest sampled would keep up with SLR. Although these forests may be accreting at rates less than SLR, they may not be immediately lost. Cahoon and Guntenspergen (2010) have proposed an elevational capital, where wetlands can continue to exist until sea level exceeds the elevation of half of their tidal range. Palau mangroves have a 2 m tidal range (Golbuu et al. 2003), a mean tidal elevation of 1 m, and a 1 m elevational capital (Lovelock et al. 2015). Using the extreme 9 mm/year SLR estimate, the interior mangrove forests in Palau have an average elevation deficit (SLR minus vertical accretion) of 5.7 ± 0.9 mm/year, while fringe mangrove forest sites have an average elevation deficit of 3.1 ± 0.1 mm/year, which means these areas would not be completely submerged for another 175 and 323 years, respectively. If SLR in Palau is indeed 9 mm/year, there may be time to implement a conservation strategy that could increase the vertical and horizontal expansion of these mangroves.

Conclusion

Palau and Vietnam mangroves appear to be keeping up with current rates of SLR, although this depends upon which SLR scenario is used in Palau. It is not clear how rates of SLR might change in the near future or how sedimentation and accretion rates will respond to changing SLR. Mangroves are resilient systems that have been able to keep up with extreme fluctuations in SLR in the past and may be able to morphologically adapt to changes in flooding regimes (Alongi 2008, 2015).

Comparison of two mangroves that differed in diversity, where Vietnam supported 35 true mangrove species and Palau supported 18 mangroves species, provided a preliminary look at how diversity may influence sedimentation rates in mangroves. The various physical and hydrological factors that influence sedimentation and accretion in mangrove forests have been identified (Krauss et al. 2014), the importance of diversity is unclear. While mangroves from only two different areas were compared, information from mangroves across a greater diversity gradient may yield additional support for (or against) this idea. This information is especially critical as many governments are investing heavily to restore once diverse mangrove forests with monotypic stands of a single mangrove species, most often *R. mucronata* (Friess et al. 2011).

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